EPA/600/R-06/155 | July 2007 | www.epa.gov/ada

EPA
United States
Environmental Protection
Agency

Wetlands and Water Quality Trading:

Review of Current Science and Economic Practices with Selected Case Studies

SCIENCE

Ground Water and Ecosystems Restoration Division, Ada, Oklahoma 74820
National Risk Management Research Laboratory
Office of Research and Development

EPA/600/R-06/155
July 2007

Wetlands and Water Quality Trading: Review of Current Science and Economic Practices With Selected Case Studies

Shane Cherry, Erika M. Britney, Lori S. Siegel, Michael J. Muscari, & Ronda L. Strauch
Prepared by Shaw Environmental Inc.
EPA Contract No. 68-C-03-097
Shaw Environmental Inc.
Cincinnati, Ohio 45212-2025

Timothy J. Canfield, Technical Monitor
U.S. Environmental Protection Agency
Office of Research and Development
National Risk Management Laboratory
Ada, Oklahoma 74820

Mary Sue McNeil, Project Officer
Ground Water and Ecosystems Restoration Division
National Risk Management Research Laboratory
Ada, Oklahoma 74820

National Risk Management Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, Ohio 45268

Notice

The U.S. Environmental Protection Agency through its Office of Research and Development funded and managed the research described here under contract No. 68-C-03-097 to Shaw Environmental Inc. It has been subjected to the Agency's peer and administrative review and has been approved for publication as an EPA document. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

All research projects making conclusions or recommendations based on environmental data and funded by the U.S. Environmental Protection Agency are required to participate in the Agency Quality Assurance Program. This project did not involve the collection or use of environmental data and, as such, did not require a Quality Assurance Project Plan.

Foreword

The U.S. Environmental Protection Agency is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the Agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. To meet this mandate, EPA's research program is providing data and technical support for solving environmental problems today and building a science knowledge base necessary to manage our ecological resources wisely, understand how pollutants affect our health, and prevent or reduce environmental risks in the future.

The National Risk Management Research Laboratory is the Agency's center for investigation of technological and management approaches for preventing and reducing risks from pollution that threatens human health and the environment. The focus of the Laboratory's research program is on methods and their cost-effectiveness for prevention and control of pollution to air, land, water, and subsurface resources; protection of water quality in public water systems; remediation of contaminated sites, sediments and ground water; prevention and control of indoor air pollution; and restoration of ecosystems. NRMRL collaborates with both public and private sector partners to foster technologies that reduce the cost of compliance and to anticipate emerging problems. NRMRL's research provides solutions to environmental problems by: developing and promoting technologies that protect and improve the environment; advancing scientific and engineering information to support regulatory and policy decisions; and providing the technical support and information transfer to ensure implementation of environmental regulations and strategies at the national, state, and community levels.

This publication has been produced as part of the Laboratory's strategic long-term research plan. It is published and made available by EPA's Office of Research and Development to assist the user community and to link researchers with their clients.

The goal of this report is to provide a review of the existing science and economic practices of using wetlands as part of water quality trading programs. This report evaluates the technical, economic, and administrative components of developing and implementing water quality trading (WQT) programs to nutrient removal is the primary focus to improve water quality. This report collates and synthesizes current literature with the goal of providing a baseline understanding of the current state of the use of wetlands in water quality trading programs. Although this document is intended to gather a significant amount of the current scientific literature available at the time of publication, it should be noted that it does not include all possible literature available on the subject due to the constantly evolving work in this area. This document should be used as a component of all the science on this subject and not considered as the sole document in this area.

Stephen G. Schmelling, Director
Ground Water and Ecosystems Restoration Division
National Risk Management Research Laboratory

Contents

Figures

Tables

Acronyms and Abbreviations

µg/L	micrograms per liter
ACWWA	Arapahoe County Water and Wastewater Authority
ADAPT	Agricultural Drainage and Pesticide Transport (model)
Association	Tar-Pamlico Basin Association
ASWCD	Ada Soil and Water Conservation District
CCBWQA	Cherry Creek Basin Water Quality Authority
BMP	best management practice
CBOD	carbonaceous biological oxygen demand
CENR	Committee on Environment and Natural Resources
cfs	cubic feet per second
CH_4	methane
CN	curve number
CO_2	carbon dioxide
CSCD	Canyon Soil and Water Conservation District
CWA	Clean Water Act
CZARA	Coastal Zone Management Act Reauthorization Amendments
DCFROI	discounted cash f ow return on investment
DSWC	Division of Soil and Water Conservation (North Carolina)
EEP	Ecosystem Enhancement Program
ETN	Environmental Trading Network
GIS	geographic information system
GWERD	Groundwater and Ecosystem Restoration Division
ICWC	Idaho Clean Water Cooperative
IDAPA	Idaho Administrative Procedures Act
IDEQ	Idaho Department of Environmental Quality
ISCC	Idaho Soil Conservation Commission
LAC	local and basin committees
lb/yr	pound(s) per year
LBR	Lower Boise River
LNBA	Lower Neuse Basin Association
mg/L	milligram(s) per liter
mgd	million gallons per day
MOU	Memorandum of Understanding
MPCA	Minnesota Pollution Control Agency
MPP	maximum purchase price
MSP	minimum selling price
N_2	nitrogen gas
N_2O	nitrous oxide
NADB	North American Wetlands for Water Quality Data Base

NANI	net anthropogenic nitrogen inputs
NBOD	nitrogenous biochemical oxygen demand
NCAC	North Carolina Administrative Code
NCDWQ	North Carolina Division of Water Quality
NCEDF	North Carolina Environmental Defense Fund
NCEMC	North Carolina Environmental Management Commission
NH_4	ammonium
NH_4-N	ammonium nitrogen
NLEW	Nitrogen Loss Evaluation Worksheet
NO_3	nitrate
NO_3-N	nitrate-nitrogen
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NPS	nonpoint source
NRCA	Neuse River Compliance Association
NRCS	Natural Resources Conservation Service
NRET	Neuse River Education Team
NRMRL	National Risk Management Research Laboratory
NSW	nutrient sensitive waters
O&M	operation and maintenance
PLAT	Phosphorus Loss Assessment Tool
PRF	Pollution Reduction Facility
PS	point source
PTRF	Pamlico-Tar River Foundation
Rahr	Rahr Malting Company
RBC	River Basin Center
SD	standard deviation
SDA	System Dynamics Analysis
Shaw	Shaw Environmental, Inc.
SISL	Surface Irrigation Soil Loss
SR-HC	Snake River-Hells Canyon
SWAT	Soil Water Assessment Tool
TD	technical directive
TKN	total Kjehldahl nitrogen
TMAL	total maximum annual load
TMDL	total maximum daily load
TN	total nitrogen
TP	total phosphorus
TSS	total suspended solids
TWDB	Treatment Wetland Database
USACE	U.S. Army Corps of Engineers
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
WQT	water quality trading
WTF	wastewater treatment facility
WWTP	wastewater treatment plant

EPA Technical Oversight Committee

Timothy J. Canfield
Ecologist
U.S. EPA, ORD, NRMRL
919 Kerr Research Drive
Ada Ok 74820

Matt Heberling
Economist
U.S. EPA, ORD, NRMRL
26 W. M. L. King Drive, MS A-130
Cincinnati, OH 45268

Kathy Hurld
Environmental Protection Specialist
U.S. EPA, OWOW, Wetlands Division
1200 Pennsylvania Avenue, NW
MCT 4502T
Washington, DC 20460

Michael Mikota
NNEMS Fellow
U.S. EPA – OWOW
1200 Pennsylvania Avenue, NW
MCT 4502T
Washington, DC 20460

Joseph P. Schubauer-Berigan
Research Ecologist
U.S. EPA, ORD, NRMRL
26 W. M. L. King Drive,
Cincinnati, OH 45268

Laurel Staley
Chief
Environmental Stressors Management Branch
U.S. EPA, ORD, NRMRL
26 W. M. L. King Drive,
Cincinnati, OH 45268

Richard Sumner
Regional Liason
U.S. EPA National Wetlands Program
200 SW 35th Street
Corvallis, OR 97333

Hale Thurston
Economist
U.S. EPA, ORD NRMRL
26 W. M. L. King Drive,
MS 499
Cincinnati, OH 45268

Cover Photo:
Clover Island - Restored wetland on a marginal agricultural field. Blacksten Wildlife Area., Kent Co. Delaware -T. Barthelmeh

Shaw Author Affiliation

Shane Cherry
Shaw Environmental and Infrastructure, Inc.
19909 120th Avenue NE, Suite 101
Bothell, WA 98011-8233
Phone: 425-218-9748
Shane.cherry@shawgrp.com

Erika M. Britney
Shaw Environmental and Infrastructure, Inc.
19909 120th Avenue NE, Suite 101
Bothell, WA 98011-8233
Phone: 425-402-3207
Erika.Britney@shawgrp.com

Lori S. Siegel
Siegel Environmental Dynamics, LLC
5 Carriage Lane
Hanover, NH 03755
Phone: 603-643-1218
lsiegel.sed@comcast.net

Michael J. Muscari
ESA Adolfson
5309 Shilshole Ave. NW, Ste. 200
Seattle, WA 98107
Phone: 206-789-9658
Fax: 206-789-9684
mmuscari@adolfson.com

Ronda L. Strauch
King County Road Services Division
King Street Center, M.S. KSC-TR-0231
201 South Jackson Street
Seattle, WA 98104-3856
Phone: 206-205-1561
Ronda.Strauch@METROKC.GOV

Executive Summary

The Groundwater and Ecosystems Restoration Division of the National Risk Management Research Laboratory serves as the U.S. Environmental Protection Agency's (USEPA) center for risk management research on ecosystem protection and restoration. It provides detailed technical guidance through Technical Directives (TD) for the technical review of papers, technical consultation, short-term project support, and field support. The current assignment for Shaw Environmental, Inc. (Shaw) addressed by this technical report is initiated by TD No. 2OA618SF and titled "Water Borne Stressor (Nutrient) Trading Program to Improve Water Quality: Science and Economic Review."

The study evaluates the technical, economic, and administrative aspects of establishing water quality trading (WQT) programs where the nutrient removal capacity of wetlands is used to improve water quality. WQT is a potentially viable approach for wastewater dischargers to cost-effectively comply with regulations and to improve water quality. The premise of WQT is that dischargers who cannot cost-efficiently reduce their eff uent loads (i.e., high cost) may buy water quality from more cost-efficient (i.e., lower cost) dischargers. Such trades may include point source (PS) dischargers, nonpoint source (NPS) dischargers, or both. This study focuses on WQT programs that allow PS-NPS trades where wetlands are used to achieve the NPS discharge reductions. The report integrates the review of published peer-reviewed literature and data sources addressing the nutrient removal function of wetlands, WQT, and the review of four case studies of existing WQT programs. Findings are used to illustrate opportunities and challenges associated with using wetlands in NPS nutrient trades. Along with any resulting research, this study should provide a technical basis for USEPA to prioritize research and publish related information resources.

The literature review addresses three concepts: (1) wetland nutrient removal, (2) trading economics, and (3) trading regulations. The case studies investigate these concepts in practice. Criteria to select the case studies included the type of program (PS-NPS); the constituent traded (nitrogen and phosphorus); implementation status; whether or not wetland construction/enhancement could be used to generate credits; and the extent to which published information was available on the program. Four case studies are evaluated: (1) Cherry Creek, Colorado; (2) Minnesota River and Rahr Malting Company (Rahr), Minnesota; (3) Lower Boise River (LBR), Idaho; and (4) Tar-Pamlico and Neuse Rivers, North Carolina.

The first category of literature review evaluates wetland nutrient removal of nitrogen and phosphorus. Constructed and natural wetlands are compared and contrasted. Both buffer downstream nutrients by storing and transforming nutrients, thereby effectively treating discharge from PSs and NPSs. The fate and transport of nutrients in wetlands is a function of dynamic biological, physical, and geochemical processes. The resulting complexities render each wetlands application unique. As such, each application warrants an evaluation of nutrient availability and the wetlands removal efficiency. Besides nutrient removal, wetlands also provide several human and ecological benefits such as f ood control, habitat for endangered and economically important species, erosion control, and recreation. Caution must be exercised, though, to avoid unintended consequences of constructed wetlands. Potential negative consequences include the loss of other productive land uses, the impairment of adjacent water bodies, danger to wildlife attracted to the wetland, inf ux of invasive plants, odor issues, and inf ux of dangerous or nuisance animals. In order for wetlands to be used for WQT, it is necessary to be able to quantify the nutrient load reduction to calculate tradable credits. Performance measurements or models/calculations of nutrient removal data can be used to quantify these credits. The lifespan of the credits, which is a function of how long the best management practice (BMP) is effective at removing nutrients, with a margin of safety, is also critical to determining the value of the wetlands for a given trade.

Economics are examined as the second category of the literature review. WQT involves buyers, sellers, and, to varying degrees, regulators. Each of these stakeholders has their own interests, concerns, challenges, and gaps. Special interests with diverse specific concerns and the general public also affect economic decisions. There are several economic trading challenges that make the risk and/or return of investing in WQT strategy unattractive to the stakeholder, thereby hindering efficient and fair deal-making and ultimately suppressing WQT. These challenges include simplified modeling of natural system impacts, expensive risk factors, high transaction costs, and undefined property rights.

Several changes to WQT program design could help overcome these obstacles by facilitating stakeholder decision-making based on an improved understanding of value and risk. While some of the changes may not necessarily increase the number of active trades, they all serve to improve the market so that trades ref ect intended goals. Measures to increase the efficiency of the trading programs would ultimately reduce the cost to develop and operate WQT exchanges. They also reduce the transaction costs of individual trades. Increasing PS compliance liability will provide a significant driver for trading. Improvements to market and non-market economic valuations of ecological services must be achieved and would help to increase the real or perceived value and opportunities NPSs can realize as a result of participating in WQT. WQT would also benefit from making tools for applying economic investment decision methods available to potential participants. Probabilistic analyses for evaluating the risk and opportunity associated with WQT should replace single-point estimate inputs, which are subject to error and bias. Probabilistic analysis would provide decision-makers with more confidence in committing capital to WQT. Finally, System Dynamics Analysis (SDA), which is a modeling process that evaluates the consequences and sequencing of complex events and phenomena inherent in many systems, would optimize the performance of the WQT market. Many of these changes simply require modifications to existing policies and have proven effective for other applications, such as business strategy development and resource management.

Finally, trading regulations are examined in the literature review. The report describes the USEPA Water Quality Trading Policy, specifically examining regulations related to wetlands. In 2003, the USEPA released its Water Quality Trading Policy to offer guidance and assistance in developing and implementing trading programs. Trading is particularly encouraged by the policy for phosphorus and nitrogen loads. The geographic area for trading programs is described by the policy as the watershed or area covered by an approved total maximum daily load (TMDL). Surplus credits are defined by the policy as constituent reductions greater than those already required by a regulation. Clear authority to trade along with unambiguous legal protection for using the purchased credits to meet established regulatory requirements is crucial for a successful WQT program. Success also mandates compliance and enforcement provisions. Programs vary based on the location and circumstances of the trading and are thus administered by the states. While strict limits on discharges drives demand for WQT, the 2007 Farm Bill will likely drive supply by compelling more NPS participation in trading. If supported by Congress, BMPs subsidized by tax dollars will become eligible to generate sellable credits.

Four case studies are evaluated according to technical, economic, and regulatory concepts. The first of these is the Cherry Creek, Colorado, case study, which is an example of a clearinghouse type of market. In 1989, the Cherry Creek Reservoir Control Regulation, listed as Regulation #72, set the stage for WQT between PS and NPS discharges of phosphorus and mandated the Cherry Creek Basin Water Quality Authority (CCBWQA) to administer the basin. The CCBWQA has been dedicated to creating and maintaining its own phosphorus reduction facilities. Furthermore, it has been committed to fostering and evaluating other BMP sources in the watershed. Three trades have occurred, one of which involved an NPS. Although these trades allowed PSs to offset some of their discharges more cost effectively, the water quality goal has yet to be achieved because the TMDL was established to accommodate growth. Nonetheless, with its f exible trading approaches and unambiguous guidelines and oversight by the CCBWQA, future success is possible.

The second case study, Rahr, in Minnesota, is an example of a sole-source offset accomplished without an established market there. In 1997, the Minnesota Pollution Control Agency (MPCA) issued to Rahr a discharge permit requiring WQT in order to satisfy the conditions of no additional oxygen-demanding discharge into the Minnesota River Basin. The permit specified acceptable BMP options, which included the three selected: critical area set-asides and wetland restoration, erosion control, and livestock exclusion. The NPS controls achieved the offsets within four years and must be maintained as long as Rahr discharges eff uent. The trades were necessary for Rahr's growth. The NPS controls implemented also resulted in other environmental and economic benefits beyond improvements to water quality. Despite the successes, limitations to the program's success exist. Instead of validating the performance of NPS controls through monitoring, reductions were evaluated by conservative assumptions, thereby requiring larger water quality improvements from the BMP projects to compensate for uncertainty, and this added expense. Furthermore, NPSs are not regulated and therefore do not have the same marketable incentive to engage in trading. Rahr will have to overcome this in the event it needs to purchase additional credits. Overall, the benefits far outweighed the limitations, rendering this trading program a success.

The third case study is the trading program in the LBR in Idaho. The Eff uent Trading Demonstration Project is a start-up program for phosphorus trading in the LBR watershed in Idaho. Although the framework of this exchange market has been established, the phosphorus TMDL has yet to be set, thereby delaying the need for trades. Nonetheless, the WQT simulation of a scenario for generating credits used sediment basins and constructed wetlands to reduce discharge. Unfortunately, high costs and use of resources to develop the trading framework hinder the program. Water rights issues discouraged buyers and sellers from participating. Potential regulation also deterred NPSs participation. Despite these issues, the participants in the demonstration project felt that the LBR framework was successful. The project highlighted issues of efficiency and uncertainties in credit calculations and BMP lifespan, and long-term fate of phosphorus removed using BMPs such as constructed wetlands.

The fourth case study comes from the Tar-Pamlico and Neuse Rivers in North Carolina. Both of these programs are based on a group cap-and-trade system and both rely on associations of PS dischargers. A nutrient offset fee must be paid for each pound of nutrient discharged beyond that collectively allowed for the association. This fee is paid to a state-administered fund for implementing BMPs to reduce the nutrient load from NPSs. Both programs successfully implemented strategies to reduce nutrient loads. The nutrient-sensitive water strategies for both basins relied heavily on public and stakeholder input. While many lessons were learned, there remain many unanswered questions regarding issues such as seasonality, nutrient removal efficiencies over time, and lifespan of the BMPs.

The literature review and case studies support a synthesis of the information regarding WQT involving NPS reductions that utilize wetlands. This synthesis summarizes the key observations of the state of WQT using wetlands based on examples provided by the case studies as well as warranted research and modifications to encourage its viability. As a cautionary note, of the more than 80 WQT programs, pilots, and simulations identified in the process of selecting the four case studies, these programs are among the longest-standing. All were developed before the USEPA issued the Water Quality Trading Policy in 2003. It is therefore recommended that some of the most recent WQT programs, for which there is currently very little published data, be evaluated to determine how and to what extent these programs are addressing the research needs and data gaps identified in this document. This said, the observations made in this document include a comparison of performance monitoring versus conservative presumption; motivations for NPS participation; effects of compliance thresholds; comparison of program structure; credit life; economic challenges to trading; and property rights and transfer of liability.

Uncertainty drives the question of performance monitoring versus conservatism, whereby high trading ratios are used to offset uncertainty. Such uncertainty derives from the dynamic, complex factors affecting wetland nutrient removal efficiency and from spatial differences between the wetlands and the PS location. Applying conservative safety factors often mitigates such uncertainty. The case studies illustrate that typically program participants presume it is more cost-effective to apply such conservatism than to directly measure the effectiveness of the constructed wetland.

WQT with NPS contributors depends on their desire to participate. The case studies demonstrate that NPS nutrient loads often exceed PS loads to a watershed. WQT programs may be used to create an economic incentive for NPSs to control their contributions by compensating them for load reductions. This is feasible in certain circumstances based on the significant difference in costs. Unfortunately, NPS contributors have a subtle disincentive to participate in trading programs in that they may lose their non-regulated status or face stricter enforcement. Stronger incentives for NPS participation call for a better understanding of nutrient loading on a watershed scale. Compliance thresholds directly affect trading attractiveness. Discharge limits must be strict enough to oblige trading, while enforcement of these limits must be credible to avoid dischargers from gaming the system instead of participating in trading.

Program structures vary considerably and include sole-source offsets, clearinghouses, and compliance associations. The various models may all be valid when executed appropriately. Questions regarding lifespan of BMPs concern the protocol beyond the expiration of credits, the temporal differences between the times of credits generation and application, and the procedure to deal with surplus credits. Economic trading challenges could suppress WQT by making the net economic value of trading less attractive than alternate compliance management strategies due to risks and uncertainties. These challenges could hinder efficient and fair deal-making because they make the risk and/or return of investing in WQT high to the buyer, the seller, or both. Lastly, the way property rights and liability transfer are addressed depends on the program. Each of the case studies manages differently the question of liability in the event of BMP failure. Lingering liability for the seller leaves unknown risk associated with trading plus additional costs, and logistics associated with monitoring BMPs implemented on the credit seller's property make WQT less attractive to PSs. Additionally, the property rights to a wetland after the credits have expired must be clear. Such doubts deter the use of constructed wetlands as a BMP in WQT programs. Long-term regulatory implications of building constructed wetlands to generate credits for WQT programs need to be clarified.

Finally, additional research recommendations within technical, economic, and regulatory categories are presented in the final section of this document. Technical research needs concern reducing uncertainty in trades involving wetlands. Several possible research topics emerge to address uncertainty in wetland performance. SDA can evaluate the complex events and phenomena inherent in many systems, thereby reducing uncertainty and quantifying risk. To address economic challenges, research must aim to determine value and risk associated with strategies that use wetlands to reduce nutrient loads. Administrative research targets regulations that promote opportunities, minimize transaction costs, formally supervise WQT implementation and compliance, assess methods to promote NPS participation, and minimize gaming risks.

WQT using wetlands is a potentially viable alternative for achieving water quality standards. This report reviews the current technical, economic, and regulatory status of this option. Based on the observed strengths and identified challenges, Shaw recommends actions to promote such programs to their fullest potential.

1.0 Introduction

The Groundwater and Ecosystems Restoration Division (GWERD) of the National Risk Management Research Laboratory (NRMRL) serves as the U.S. Environmental Protection Agency's (USEPA) center for risk management research on ecosystem protection and restoration, focusing its efforts on studies to assess and enhance the ability of terrestrial and aquatic ecosystems to support and maintain water quality, support native species of plants and animals, and to provide ecological services on a watershed scale. Shaw Environmental, Inc. (Shaw) receives detailed technical guidance and direction from NRMRL/GWERD in the form of Technical Directives (TD) for the areas of technical review of papers, technical consultation, short-term project support, and field support. The current assignment addressed by this technical report is initiated by TD No. 2OA618SF and titled "Water Borne Stressor (Nutrient) Trading Program to Improve Water Quality: Science and Economic Review."

The relative importance of point sources (PS) and nonpoint sources (NPS) of nutrients varies from watershed to watershed. However, according to an agriculture handbook published by the U.S. Department of Agriculture (USDA), "national-scale water quality assessments strongly suggest that agriculture is a leading source of remaining water quality problems" (Heimlich, 2003). Nutrient inputs into the waters of the United States continue to be one of the major reasons that water bodies do not meet their designated uses as defined under the Clean Water Act (CWA; Federal Water Pollution Control Act Amendments of 1972, later amended in 1977). USEPA instituted a Water Quality Trading Policy to encourage trading as an innovative way of meeting water quality goals within a watershed context (USEPA, 2003a). The policy is based on the idea that different sources within a watershed may face drastically different costs to control the same constituent. Trading programs, which have proved to be very successful in meeting air quality standards, allow facilities facing higher discharge control costs to meet their regulatory obligations by purchasing environmentally equivalent, or superior, reductions from another source at lower cost than they would incur by installing additional controls. To date, this policy has been implemented to a limited extent for PS-PS trading. There is a great deal of interest in increasing the implementation of this policy for PS-NPS trading, particularly through the use of wetlands (Schubauer-Berigan, 2005; Raffini and Robertson, 2005), but there appear to be a number of possible gaps in the available scientific and economic knowledge needed to implement such trading as part of a regulatory program.

1.1 What is Water Quality Trading?

Water quality trading (WQT) is a voluntary alternative for achieving regulatory compliance with water quality standards. It is a program whereby parties can meet their discharge allowances by trading with each other. Although it has been available for over two decades, this option is just recently garnering more attention. In WQT, cost-inefficient dischargers[1] buy water quality credits from cost-efficient dischargers, who have earned credits by voluntarily implementing best management practices (BMPs) for nutrient control. By trading credits, the overall cost of achieving nutrient reduction is minimized. In an efficient market, WQT leads to lowest-cost nutrient reduction.

An established market or exchange provides the structure for the WQT transactions. The regulator or some other entity plays a third-party role in the market, protecting the interests of the public by ensuring that trading maintains or improves water quality and does not lead to degradation of the environment.

Overall, economists, regulators, dischargers, environmentalists, and other stakeholders have advocated WQT as a way to use market-based solutions to reduce the cost of complying with water quality discharge limits. The approach provides PSs with alternatives for controlling discharges with less regulation, less cost, and accelerated compliance. The f exibility afforded by WQT that includes NPSs can create ecological value without increasing natural resource risk. Regulatory oversight controls the process.

1.2 Report Overview

The initial work plan for the study included a broad assessment of published literature pertaining to WQT programs that include NPS trades. As the study progressed, collaboration between the study sponsors and the authors focused the scope of the study on the use of wetlands as an NPS control to reduce nutrient loads and create credits for trade.

1 *In this document, "discharger" is a term used to refer to both PSs and NPSs whose discharge is due to human influences.*

The study evaluates the technical, economic, and administrative aspects of establishing WQT programs that can use and have used wetlands to generate credits for NPS trades. The evaluation relies upon a review of technical literature combined with selected case studies. The literature review and case studies are used to identify critical scientific and economic knowledge gaps that would impede the implementation of a WQT program including both PSs and NPSs. Although examples from several case studies facilitate specific points in the wetlands, economics, and regulatory reviews, this report considers the four programs included as case studies to illustrate the current state of practice of using wetlands in WQT programs. Although the programs described in the case studies are not markets, they are illustrative of important aspects of WQT involving wetlands. Based on the synthesis of this work, the USEPA will be able to develop a plan to research gaps regarding using wetlands to generate NPS credits in WQT. Addressing these gaps will provide insight towards assessing the feasibility of such programs and identify factors to opt for certain approaches.

2.0 Methods for Identifying Technical and Economic Analysis Needs

The current investigation combines a review of published literature and a case study analysis to establish and evaluate the state-of-the-art in WQT programs. By evaluating existing regionally focused WQT programs, the study identifies data and knowledge gaps and recommends research to address them. Ultimately, this review and any resulting research would enable USEPA to publish technical information for using wetlands in PS-NPS WQT programs. This study integrates two primary components: (1) review of published peer-reviewed literature and data sources addressing WQT and wetlands nutrient removal functions, and (2) review of four case studies of existing WQT programs. The literature review and case study analysis results are used to assess opportunities and potential pitfalls associated with using wetlands in NPS nutrient trades.

Shaw collaborated with USEPA to develop a list of critical questions to screen and compile relevant literature and other available sources of information for the area of WQT programs for nutrients. The primary sources of information are derived from published peer-reviewed literature, including articles from scientific and economic journals, conference proceedings, and books. Other information sources include relevant federal and state regulations. Information gained from secondary and non-peer-reviewed sources, including conference proceedings, workshops, white papers, fact sheets, web sites, etc., is used to illustrate the level of interest in WQT.

The literature review will produce a list of issues pertaining to the successful operation of WQT programs along with published data and a bibliography addressing each of these issues. The association of issues and available data will illustrate the nature and extent of data and knowledge gaps.

2.1 Literature Search Methodology

The literature review was conducted as an iterative process by listing issues to inform an initial literature search. Candidate source documents were compiled, screened according to the critical questions, and then sorted according to subject. A combination of methods was used to identify documents included in the literature review. These methods included use of internet search engines; personal communications with experts, such as the contact people for each of the case studies; agency internet sites, such as the web pages for individual WQT programs; reviewer comments; and references contained in publications already identified. A complete list of all documents identified during the literature review is composed as an annotated bibliography in Appendix A.

The following internet search engines and search terms were used to identify relevant documents.

Table 2-1. Internet Search Engines and Search Criteria

Search engines	Search terms	Date limits
Agricola http://agricola.nal.usda.gov/webvoy.htm	Wetland and nitrogen, wetland and treatment, wetland and constructed, WQT, assess WQT, assess nutrient trade, assess nutrient credit, assess nutrient models, validate nutrient models, compare nutrient models, nutrient trading	2000 to January 2006
Ecological Society of America http://www.esajournals.org/esaonline/?request=search-simple	Wetlands, nitrogen, nutrients, WQT, nutrient trading	None
Elsevier http://www.elsevier.com	Minnesota Pollution Control Agency (MPCA), Rahr Malting Company (Rahr), Cherry Creek, publications, WQT, total maximum daily loads (TMDL), equivalence, wetlands AND WQT, specific author names, nutrient trading	None
Google Scholar http://scholar.google.com/	WQT, NPS trading, pollutant trading programs, North Carolina case study specific terms: Tar-Pamlico, Neuse, Trading Program, water quality, wetlands, specific author names, TMDL, nutrient trading	None

Search engines	Search terms	Date limits
Google http://www.google.com	WQT, assess WQT, assess nutrient trade, assess nutrient credit, assess nutrient models, validate nutrient models, compare nutrient models, MPCA, Rahr, Cherry Creek, WQT, Idaho DEQ, Idaho Soil Conservation Commission (ISCC), Lower Boise River (LBR), nitrogen, phosphorus, TMDL, equivalence, wetlands AND WQT, specific author names, nutrient trading	None
PubMed database http://www.ncbi.nlm.nih.gov/entrez/query.fcgi?CMD=search&DB=pubmed	Wetlands, nitrogen, nutrients, WQT, NPS trading, nutrient trading	None
Science Direct http://www.sciencedirect.com/	Wetlands, nitrogen, nutrients, assess WQT, assess nutrient trade, assess nutrient credit, assess nutrient models, validate nutrient models, compare nutrient models, WQT, NPS trading, nutrient trading	None
State environmental organization search engines	MPCA, Rahr, Cherry Creek, publications, WQT, TMDL, equivalence, wetlands AND WQT, specific author names, NPS pollution, nutrient trading	None
Wetlands website (SWS journal) http://www.sws.org/wetlands/	Wetlands, nitrogen, nutrients, WQT nutrient trading	None
Environmental Trading Network (ETN) http://www.envtn.org/index.htm	Workshops 2nd National Water Quality Trading Conference, held May 23-25, 2006 in Pittsburgh. (http://www.envtn.org/WQTconf_agenda.htm) Environmental Credits Generated Through Land-Use Changes: Challenges and Approaches held March 8-9, 2006 in Baltimore. http://www.envtn.org/LBcreditsworkshop/agenda.htm	None
Environmental Law Institute http://www2.eli.org/index.cfm	Workshop National Forum on Synergies Between Water Quality Trading and Wetlands Mitigation Banking held July 11-12, 2005 in Washington, DC. http://www2.eli.org/research/wqt_main.htm.	None

2.2 *Literature Review Questions*

Literature screening criteria are grouped into three categories: Level 1 – Preliminary Screening Questions for Identification of Case Studies; Level 2 – Case Study Analysis Questions; and Level 3 – General “State of the Art” Questions. The case studies are used to address the Level 1 and 2 questions. The Level 3 group of questions was created with the recognition that the case studies may not be able to directly answer these questions.

2.2.1 Level 1 – Preliminary Screening Questions for Selection of Case Studies

1. Are there any published case studies of WQT programs within the United States or other countries?
2. How far (spatially) are the benefits of a local nutrient load reduction realized within a water body? How does this vary for different designated water uses? How does this vary between watersheds or different water body types (e.g., estuary, river, lake) with distinct hydrologic, geologic, and ecologic conditions? How can appropriate geographic trading areas be established?
3. To what extent does seasonal variability need to be accounted for in trading programs?
4. What are the economic factors that drive the feasibility of various nutrient load reduction measures? How do these factors vary depending on location and watershed conditions?
5. How should the cap for nutrient concentrations in water bodies be defined, especially in multi-state waters? How should a baseline be established?
6. What factors determine the effectiveness of wetlands for reducing or removing nutrients from surface water?
7. If the price for a nutrient loads reduction credit from an NPS is fixed (e.g., $/lb) within a trading program, how are agencies determining the credit price?

8. How can nutrient reductions from NPSs be quantified? How is “effectiveness” of various management practices measured and documented? How can a reduction be measured after a management practice has been implemented? How can the initial NPS nutrient load be quantified?
9. What are the various ways that trading is being managed? What are the advantages (or drawbacks) of each management approach? To what extent is the management approach dependent on program scale or types of water body included in the program?
10. For multi-state (multi-jurisdiction) trading programs, how can legal authority be established?

2.2.2 Level 2 – Case Study Analysis Questions

1. What have been the key drivers for the implementation of a WQT focused on nutrients, or other environmental performance trading programs (such as air quality and wetland mitigation)?
2. What factors contribute to the success of active WQT programs or limit their effectiveness?
3. What type of institutional framework can provide accountability of NPSs? How can compliance with regulations be assured and enforced?
4. What role should environmental groups have in the planning and implementation process? How much public participation is appropriate?
5. What is public perception of water-borne stressor (nutrient) trading programs? Are there organizations opposed to this type of program?

2.2.3 Level 3 – General “State of the Art” Questions

1. What federal regulations and guidance documents address WQT?
2. What state regulations and guidance documents address WQT?
3. Which states have active WQT programs?

2.3 Case Study Selection

A few basic selection criteria were used to choose case studies from the list of existing WQT programs compiled in Table 2-2. The selection criteria include type of program (PSs and NPSs); constituent traded (nitrogen and phosphorus); implementation status (the program needed to be fully developed); whether or not wetland construction/enhancement could be used to generate credits; geographic distribution; and the availability of published literature. Four case studies were selected:

1. Cherry Creek, Colorado
2. Minnesota River and Rahr, Minnesota
3. LBR, Idaho
4. Tar-Pamlico River and Neuse River, North Carolina

These case studies were selected to represent programs in different regions of the country in an attempt to illustrate region-specific issues or limitations on feasibility if they exist. To the extent possible, case studies were selected to include distinct watershed types varying in scale, topography, land use distribution, and proximity to coastal waters. Market structure was not a selection criteria; the Cherry Creek and North Carolina programs may not fit the definition of a “true market” because purchase and sale of credits occur via a clearing house. In addition, water quality credits in the North Carolina program function more like an exceedance tax than trades within a market. The need for published literature on the WQT program was also a factor that shaped this analysis. Of the more than 80 WQT programs, pilots, and simulations identified in the process of selecting the four case studies, these programs are among the longest-standing. All were developed before the USEPA Water Quality Trading Policy was published in 2003, although these programs are far from static. As a result, it is likely that some of the newest programs have already been able to apply lessons learned from the programs in their design and implementation.

The collective results of the case studies combined with the results of the literature review are used to identify common lessons learned, successes and failures, and variations in key issues related to geography, watershed scale, land use, and any other factors observed to affect the success of the case study trading programs.

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
1. Montgomery Water Works and Sanitary Sewer Board	Coosa River	AL	Undefined - nutrients	10	No	No	No – Initial development
2. City of Santa Rosa	Russian River	CA	Undefined - nutrients	10	No	No	No – No trading
3. Grassland Area Tradable Loads Program	San Joaquin River	CA	Selenium	10, 261	Yes	No	No – Selenium trading
4. Lake Tahoe Water Quality Trading Strategy	Lake Tahoe	CA & NV	Nutrients and sediment	10	No	Yes – wetland controls, wetland type not specified	No – Initial planning stages
5. Sacramento Regional County Sanitation District's Mercury Offset Program	Sacramento Area	CA	Mercury	10	No	No	No – Mercury trading
6. San Francisco Bay Mercury Offset Program	San Francisco Bay	CA	Mercury	10	No	No	No – Mercury trading
7. Bear Creek Trading Program	Bear Creek Reservoir	CO	Phosphorus	10	No	No	No – Point-to-point
8. Boulder Creek Trading Program	Boulder Creek	CO	Nitrogen	10	No	Yes – habitat restoration and constructed wetlands (riparian)	No – Limited information available
9. Chatfield Reservoir Trading Program	Chatfield Reservoir	CO	Phosphorus	10, 114	Yes	? – BMPs for stream bank restoration and stormwater runoff	No – Limited information available
10. Cherry Creek Basin Trading Program	Cherry Creek Reservoir	CO	Phosphorus	1, 10, 11, 150, 225, 293	Yes	Yes – constructed wetlands, (riparian)	Yes – One of the original projects involved creation of a wetland. Credits established on case-by-case basis.
11. Clear Creek Trading Program	Clear Creek	CO	Heavy Metals	10, 181	Yes	No	No – Mine discharge
12. Lower Colorado River	Colorado River	CO	Selenium	10	No	No	No – Selenium trading
13. Lake Dillon Trading Program	Dillon Reservoir	CO	Phosphorus	10, 181, 236,149	Yes	No	No – Wetlands not used

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
14. Long Island Sound Trading Program	Long Island Sound	CT	Nitrogen	1, 10, 174	Yes	No	No – Point-to-point
15. Blue Plains Wastewater Treatment Plant (WWTP) Credit Creation	Chesapeake Bay	VA	Nitrogen	10		No	No – Point-to-point
16. Tampa Bay Cooperative Nitrogen Management	Tampa Bay	FL	Nitrogen	10	Yes	No	No – Wetlands not used
17. Lake Allatoona Watershed Phosphorus Trading Program	Lake Allatoona Watershed	GA	Phosphorus	10, 195, 215	Yes	? – type not specified	No – In development
18. Cargill and Ajinomoto Plants Permit Flexibility	Des Moines River	IA	Ammonia and CBOD		No		No – Limited information available
19. Bear River Basin	Bear River	ID, WY, UT	Phosphorus	10	No	?	No – In development
20. Lower Boise River Eff uent Trading Demonstration Project	Lower Boise River	ID	Phosphorus	1, 10, 174, 270, 236	Yes	Yes – Constructed wetlands, wetland type not specified	Yes – Constructed wetlands on approved BMP list, which also identified life span.
21. Mid-Snake River Demonstration Project & Development of Idaho Water Pollutant Trading Requirements	Mid-Snake River	ID	Phosphorus		No	No	No – Limited information available
22. Lake Erie Land Company/Little Calumet River	Little Calumet River	IN	Undefined	10	No	No	No – Initial development
23. Illinois Pretreatment Trading Program	IL waters	IL	Multiple	10	No	No	No – Point-to-point
24. Piasa Creek Watershed Project: Water Quality Trading - PS for NPS	Piasa Creek Watershed	IL	Sediment	10, 36	Yes	? – sed. ctrl. structures	No – No wetlands
25. Monocacy River	Monocacy River	MD	Undefined	10	No	No	No – Initial development
26. St. Martin's River Watershed	St. Martin's River Watershed	MD	Undefined	10	No	No	No – Initial development
27. Wicomico River	Wicomico River	MD	Undefined	10	No	No	No – Initial development
28. Charles River Flow Trading Program	Charles River	MA	Water f ow	10, 98	Yes		No – Water f ow trading

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
29. Edgartown WWTP	Edgartown River	MA	Nitrogen	10	No	No	No – Sewer/septic only
30. Falmouth WWTP	Falmouth Harbor	MA	Nitrogen	10	No	No	No – Sewer/septic only
31. Massachusetts Estuaries Project	Popponesset Bay, Three Bays and Warham Bay and Agawam River	MA	Nitrogen	10	No	Yes – wetland type not specified	No – Limited information available
32. Nashua River	Nashua River	MA	Phosphorus	10	No	No	No – Initial development
33. Town of Acton POTW	Assabet River	MA	Phosphorus	10	No	No	No – No trades
34. Specialty Minerals, Inc. in Town of Adams	Hoosic River	MA	Temperature	10	No	No	No – Temperature trading
35. Wayland Business Center Treatment Plant Permit	Sudbury River	MA	Phosphorus	10	No	No	No – Sewer/septic only
36. Maryland WQT Policy	Chesapeake Bay, other MD waters	MD	Phosphorus and nitrogen	10	Yes	No	No – Wetlands not used
37. Kalamazoo River Water Quality Trading Demonstration	Kalamazoo River, Lake Allegan	MI	Phosphorus	10, 261, 233, 236, 204, 226	Yes	agri. BMPs	No – Wetlands not used
38. Michigan Water Quality Trading Rule Development	MI waters	MI	Phosphorus and nitrogen	1, 10, 174, 236	Yes	No	No – Regs. only
39. Saginaw River Basin	Saginaw River Basin	MI	Nutrients and sediment	236	Yes	? – type not specified	No – No trades
40. Minnesota River Basin	Minnesota River	MN	Phosphorus	10, 63, 174, 233, 236,105, 242, 252	Yes	Yes – BMP and wetland type not specified	Yes – High volume of trades
41. Minnesota River WQT Study	Minnesota River	MN	Phosphorus	10	Yes	No	No – Point-to-point
42. Rahr Permit (lower Minnesota River)	Minnesota River	MN	Phosphorus, nitrogen, $CBOD_5$ and sediment	1, 10, 261, 133, 193, 194	Yes	Yes – restored riparian wetlands	Yes – Specifically tied to National Pollutant Discharge Elimination System (NPDES) permit requirements

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
43. Southern Minnesota Beet Sugar Cooperative Plant Permit	Minnesota River	MN	Phosphorus	1, 10	Yes	Yes – constructed wetlands, wetland type not specified	No – Limited information available
44. Mississippi River/Gulf of Mexico	Mississippi River/ Gulf of Mexico	MS	Phosphorus and nitrogen	233, 68, 30, 31, 74, 76	Yes	Yes – wetland restoration, wetland type not specified	Yes – But in concept stage of development
45. Chesapeake Bay WQT Program	Chesapeake Bay	Multiple states	Phosphorus and nitrogen	10	Yes	No	No – Wetlands not used
46. Great Lakes Trading Network	Great Lakes	multi-defined by individual programs	Undefined		Yes	No	No – See Kalamazoo
47. Cape Fear River Basin	Cape Fear River	NC	Undefined	10		Yes – wetland restoration and constructed wetland, wetland type not specified	No – Initial planning stages
48. Neuse River Nutrient Sensitive Water (NSW) Management Strategy	Neuse River Estuary	NC	Nitrogen	10, 174, 96,46, 129, 132, 46, 129, 132	Yes	Yes – wetland restoration and constructed wetland (riparian)	Yes – Cooperative - PS purchase credits, central agency (North Carolina Wetland Restoration Fund) allocates funds to projects. Nutrient offset payments targeted toward restoration of wetlands and riparian areas within the Neuse River Basin.
49. Tar-Pamlico Nutrient Reduction Trading Program	Pamlico River Estuary	NC	Phosphorus and nitrogen	1, 10, 261, 178, 96, 157, 236, 52, 112, 116, 128, 130, 131, 151, 178	Yes	Yes – emphasis on agricultural BMPs, wetland restoration and constructed wetland (riparian)	Yes – One of the oldest trading programs in the US. Cooperative - PS purchase credits, central agency allocates funds to projects.
50. Passaic Valley Sewerage Commission Eff u-ent Trading Program	Hudson River	NJ	Heavy Metals	10	No	No	No – Dissolved metal trading

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
51. Truckee River Water Rights and Offset Program	Truckee River	NV	Phosphorus, nitrogen, or total dissolved solids	10	No	No	No – Wetlands not used
52. East River	East River	NY	Nitrogen	10, 166	No	No	No – Point-to-point
53. New York City Watershed Phosphorus Offset Pilot Programs	Hudson River	NY	Phosphorus	10	Yes	Yes – wetland restoration, type not specified	No – Limited information available
54. Greater Miami River Watershed Trading Pilot Program	Greater Miami River Watershed	OH	Phosphorus and nitrogen	10	Yes	? – see types of agricultural BMPs	No – Limited information available
55. Clermont County Project	Little Miami River, Harsha Reservoir	OH	Phosphorus, nitrogen or total dissolved solids	10	No	No	No – Wetlands not used
56. Ohio River Basin Trading	Ohio River Basin	OH – Multi-state	Nutrients	10	No	No	No – Initial development
57. Shepard Creek (tributary to Mill Creek)	Shepard Creek	OH	Peak stormwater f ows	256	Yes	No	No – Stormwater retention
58. Honey Creek Watershed	Honey Creek	OH	Phosphorus	10	No	No	No – BMP case study, not trading
59. Lower North Canadian River	Lower North Canadian River	OK	Undefined	10	No	No	No – Feasibility study
60. Tualatin River Watershed NPDES Permits	Tualatin River Watershed	OR	Temperature	10	No	No	No – Riparian restoration
61. Conestoga River	Conestoga River	PA	Phosphorus and nitrogen	10	No	No	No – Wetlands not used
62. Pennsylvania Water-based Trading Simulations	Delaware River, Moshanon Creek, Swatara Creek and Spring Creek	PA	Multiple	10	No	No	No – Simulation, not program
63. Pennsylvania Multimedia Training Registry	State-wide	PA	Phosphorus and nitrogen	10	No	No	No – Wetlands not used

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
64. Eff uent Trading Program	Providence and Seekonk Rivers, Rhode Island	RI	Salinity	10	No	No	No – Salt trading
65. Boone Reservoir	Boone Reservoir	Total nitrogen (TN)	Phosphorus, nitrogen and BOD	10	No	No	No – No program developed
66. Colonial Soil and Water Conservation District	Lower James River	VA	Nutrients and sediment	10	No	No	No – Planning
67. Henry County Public Service Authority and City of Martinsville Agreement	Smith River	VA	Total dissolved solids	10	No	No	No – Point-to-point
68. Virginia Water Quality Improvement Act and Tributary Strategy	Chesapeake Bay, other VA waters	VA	Phosphorus and nitrogen	10	No	No	No – Planning
69. Chehalis River	Chehalis River	WA	Undefined	10	No	No	No – Not implemented
70. Puyallup River	Puyallup River	WA	Ammonia and BOD	10	No	No	No – No trades
71. Yakima River	Yakima River	WA	Water f ow	10	No	No	No – Water quantity
72. Fox-Wolf Basin Watershed Pilot Trading Program	Green Bay	WI	Phosphorus	10, 178	Yes	No	No – Point-to-point, nonpoint not defined
73. Red Cedar River Pilot Trading Program	Tainter Lake	WI	Phosphorus	10	Yes	No	No – Wetlands not used
74. Rock River Basin Pilot Trading Program	Rock River Basin	WI	Phosphorus	10, 236	Yes	Yes – wetland restoration (return farmland to wetland), type of wetland not specified	No – Limited information available
75. Wisconsin Eff uent Trading Rule Development	WI waters	WI	Phosphorus	10	No	No	No – Wetlands not used
76. West Virginia Trading Framework	State-wide	WV	Multiple	10	No	? – some info on wetlands, type not specified	No – Concept stage

Table 2-2. Waterborne Stressor (Nutrient) Trading Programs

Project	Water body	State	Constituent	Ref. (doc#)	Program-specific papers	Wetlands used in trading?	Candidate study (why)
77. Cacapon/Lost River	Lost River	WV	Undefined	10	No	No	No – Feasibility study
78. Cheat River, West Virginia	Cheat River	WV	Heavy Metals and acidity	10	No	No	No – Concept stage
79. Hunter River Salinity Trading, USEPA Department of Environment and Conservation	Hunter River	Australia	Salinity		Yes	No	No – Salinity trading
80. Dutch Nutrient Quota System	Country-wide	Netherlands	Nutrients		Yes	No	No – Livestock production
81. South Nation River Watershed	South Nation River	Ontario, Canada	Phosphorus	34, 273, 272, 290	Yes	?	No – Focus on agriculture BMPs and riparian stabilization
82. Kaoping River Basin	Kaoping River Basin, Taiwan	Taiwan	Multiple	70, 75	No	No	No – Limited information available

Candidates for case studies are highlighted in green.

CBOD = carbonaceous biological oxygen demand.

3.0 Literature Review – Wetland Nutrient Removal

The utility of wetlands in managing nutrient loads and their historical, current, and anticipated future implications in WQT warrant focused review. Numerous studies or summaries of studies have investigated the function of wetlands in the removal of pollutants, including high levels of nutrients (USEPA, 2005a; Fisher and Acreman, 2004; Mitsch and Gosselink, 2000; Hunt and Poach, 2001; Kadlec and Knight, 1996; USEPA, 1999; USEPA, 1993a; Cooper and Findlater, 1990). Results from these studies have been summarized and used to guide the development of constructed wetlands to treat water high in nitrogen and phosphorus (Kadlec and Knight, 1996). This review does not attempt to re-summarize these studies, but references them for readers who desire more information. Rather, this review summarized information on the nutrient removal function of wetlands specifically applicable to WQT.

A bibliography of published documents regarding constructed wetlands was compiled by USDA staff from the Ecological Sciences Division of the Natural Resources Conservation Service (NRCS) and the Water Quality Information Center at the National Agricultural Library. The references were acquired in part through searches of the AGRICOLA database. The bibliography has been updated several times, most recently in June of 2000, and contains hundreds of entries, many with abstracts (USDA, 2000). An annotated bibliography of urban stormwater and nonpoint nutrient control was conducted by the Washington State Department of Ecology in 1986 and updated in 1991. The review was conducted to determine the extent of information available on the long-term ecological impacts of stormwater on wetlands and on the ability of wetlands to improve the water quality of urban stormwater (Stockdale, 1991).

Both constructed and natural wetlands function to buffer downstream nutrients by storing and transforming nutrients, which are gradually released downstream (DeBusk, 1999). Consequently, wetlands have been considered an effective means to treat PSs and NPSs of nutrients and improve water quality in downstream lakes and rivers. The benefits of using wetlands to treat NPSs of pollutants include the ability to operate under a wide range of hydraulic loads, provide internal water storage capacities, and remove or transform contaminants (Dierberg *et al*., 2002).

3.1 Wetland Removal of Nitrogen and Phosphorus - Technical Overview

Nutrients enter wetlands through various geologic, biologic, and hydrologic pathways; however, hydrologic inputs generally dominate elemental inputs into wetlands. The cycling of nutrients in wetlands has been extensively described and studied (Mitsch and Gosselink, 2000). Inundation, water level f uctuations, and biota result in both aerobic and anaerobic processes within the water column and wetland soils. These processes allow the transformation of nutrients like nitrogen and phosphorus as they interact with the biogeochemistry of the wetland environment.

Wetlands function to remove phosphorus through sedimentation, plant uptake, organic matter accumulation, immobilization, and soil sorption. Nitrogen is removed in wetlands by filtration, sedimentation, uptake by plants and microorganisms, adsorption, nitrification, denitrification, and volatilization. Gaseous losses of nitrogen through denitrification are generally the most significant nitrogen removal mechanism in natural as well as constructed freshwater wetlands (DeBusk, 1999; Bowden, 1987; Faulkner and Richardson, 1989).

A description of inputs, outputs, and internal cycling of nutrients in wetlands can be described by chemical mass balances. These mass balances for wetlands have been developed and discussed by others to describe the functions of wetlands in nutrient production and cycling. Literature reviews of this subject have been provided by DeBusk (1999), Nixon and Lee (1986), Johnston *et al*. (1990), and Johnston (1991). However, few investigators have developed a complete mass balance for wetlands that includes measurement of all the nutrient pathways, sources, and sinks. Despite this lack of comprehensive study, some generalizations have been made (Mitsch and Gosselink, 2000).

The function of wetlands as sources, sinks, and transformers of nutrients depends on the wetland type, hydrologic condition, and the length of time the wetland is subjected to nutrient loading. Wetlands have been shown to be sinks or storage places for nitrogen and phosphorus, although not all wetlands exhibit this trait. One study found seasonal and permanent swamps had a net export of organic matter. Most of the inorganic phosphorus (60 to 90 percent) was retained, but there was a net release of nitrates, probably associated with the net export of organic matter (Mwanuzi *et al*., 2003). The location and chemical form of nutrients change within wetlands during the exchange of water and sediment as well as during plant uptake and decomposition (Atlas and Bartha, 1981). The availability of nutrients and the extent to which biogeochemical processes function affect the intracycling of nutrients and the productivity in wetlands. The function of wetlands is closely related to adjoining land and water bodies; changes upgradient of a wetland will affect processes occurring within the wetland. For example, the depth of an adjoining water body or the conveyance capacity of the outlet stream are likely to modulate functions such as depth and storage capacity of natural wetlands (Kadlec and Knight, 1996).

The productivity of wetlands is also directly correlated with nutrient input and transformation. Thus, the ability of wetlands to store and transform nutrients is directly connected to the amount of nutrients available for storage and transformation. However, this ability is not limitless, and once storage and transformation capacity is reached, excess nutrients leave the wetland through atmospheric, surface, and subsurface outflows (Mitsch and Gosselink, 2000). If long-term nutrient removal is an objective of a constructed wetland, significant maintenance up to and including re-construction may be necessary, although expecting a constructed wetland to perform this function in perpetuity is likely ecologically and economically unrealistic at best, and not reasonably feasible at worst.

Although several generalizations can be made regarding the function of wetlands as sources, sinks, and transformers of nutrients, the complex and unique situation revolving around each wetland limits the application of generalizations. Wetlands can be a sink for a form of nitrogen at one moment in time and a source for the same nitrogen element at another time. Generalizations are also hampered by inconsistent study results and by the variety and imprecision of approaches to measuring nutrient fluxes in wetlands. There is little consensus in the literature about nitrogen and phosphorus fate in wetlands. A few chemical imbalances have been studied and described, but a complete mass balance for wetlands has yet to be developed (Mitsch and Gosselink, 2000). Furthermore, there has been a terrestrial-biased (i.e., applying processes found in uplands) approach in wetland research, especially regarding vegetation and productivity, that limits the understanding and employment of soil and microbial processes specific to wetlands in nutrient reduction (Wetzel, 2001; Johnston, 1991).

The chemical transformation of nitrogen and phosphorus is important to understanding how wetlands perform in nutrient removal and sequestration. Inorganic and organic nitrogen and phosphorus enter wetlands through water inputs such as overland runoff, outfall pipes, groundwater, and to a lesser degree rainfall. The inorganic and organic forms are transformed or stored in the water, soil, and biota through several processes, including nitrification, denitrification, ammonification, diffusion, plant uptake, litterfall, decomposition, adsorption, precipitation, sedimentation, volatilization, and peat accretion (DeBusk, 1999). Following transformation and storage, both inorganic and organic forms of nitrogen and phosphorus exit the wetland in water outflows or by gaseous states such as nitrogen gas (N_2). Other gases are emitted from wetlands, including carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4), which are produced under highly reduced conditions (Mitsch and Gosselink, 2000).

To use wetlands to reduce nutrients from water before the flows enter downstream water bodies, the amount of nutrients in the wetland outflow needs to be less than in the wetland inflow, and the reduction must be measurable. The USEPA found that sequential nitrogen transformation within wetlands used to treat water quality results in a unidirectional shift of elevated total and organic nitrogen forms to oxidized or gaseous nitrogen forms (USEPA, 1999). In addition, plant detritus provides long-term storage of nitrogen in wetlands, and a portion of this nitrogen can eventually become available for nutrient cycling following decomposition, which can take from months to many years (Kadlec and Knight, 1996). A summary of data collected in North American Wetlands for Water Quality Data Base (NADB) found that free water surface wetlands on an annual mean average removed 61 percent of the total phosphorus (TP) in inflow water with a standard deviation (SD) of 30 percent (USEPA, 1999). An approach to control the impacts of elevated nutrients is for the nutrients to be in a form not readily available to biotic organisms such as algae, which consume oxygen during uptake of nutrients. For example, phosphorus chemically bound to minerals (e.g., iron, aluminum, calcium, and organic compounds) is not as readily available as dissolved phosphorus to algae or plants, but represents a long-term source of phosphorus in a water system (NRCS, 2001).

One of the key environmental drivers in nutrient transformation is inundation. Inundation affects the oxygen content of the soil and produces anaerobic conditions, although the near-surface soil tends to retain an oxidized layer due to the proximity to the water column, oxygen translocation within rooted plants, and microbial activity (Tanner, 2001a). Some studies have found oxygen availability to the sediment was the greatest limiting factor for nitrification (White and Reddy, 2003). Oxidation affects the reduction of elements such as iron, resulting in a brownish-red color at the soil surface compared to the bluish-gray color of reduced sediments dominated by ferrous iron. Subsurface systems have been found to display marginal or negative nitrogen removal because of the lack of oxygen (USEPA, 1993a). Inundation also affects pH and redox potential, which influences the rates of nutrient transformation (Mitsch and Gosselink, 2000).

The results from studies on nutrient removal have shown inconsistencies in amount and efficiency of nutrient removal. For example, results from an experimental constructed wetland showed that nutrient removal was primarily the result of plant uptake and harvesting (15 percent of TN input, 10 percent of TP input). Other processes had a relatively minor contribution: denitrification (8 percent of TN input), sedimentation and accumulation of organic matter in the soil (7 percent of TN input, 14 percent of TP input) (Meuleman *et al*., 2003). Other studies have shown that denitrification is one of the more important mechanisms for removing nitrogen in wetlands. Nitrogen removal from septage with high solids concentration resulted from sedimentation of waste solids (57.6 percent), denitrification (40.9 percent), and direct uptake by plants (0.5 percent) of the total inf uent nitrogen (Hamersley *et al.,* 2001). Recent studies show a wide range of nutrient removal efficiency values. Studies of constructed surface f ow wetlands in Norway found nitrogen removal efficiencies between 3 and 15 percent, due to high hydraulic load and low temperatures (Braskerud, 2002). In constructed horizontal reed bed wetlands in Germany, more than 90 percent removal of TN and phosphorus was achieved (Luederitz *et al*., 2001). A compilation of data from 60 studies of 57 natural wetlands in 16 countries showed the mean percent change in nutrient load between water entering and exiting the wetlands was 67 percent (SD of 27 percent) for nitrogen and 58 percent (SD of 23 percent) for phosphorus (Fisher and Acreman, 2004).

One of the primary ways nutrients are removed from inf ow waters is through storage within the wetlands, typically within soil, organic matter, or biota. For example, phosphorus is stored in wetlands in the soil by adsorption (i.e., surface accumulation) with sediment particles and precipitation with other compounds, within peat and plant litterfall, and in living plant and animal biomass (e.g., bacteria, algae, and vascular macrophytes). Sediment containing high organic matter accumulated twice the nitrogen (Tanner, 2001b) and six times the phosphorus (Tanner, *et al*., 1998) of live and dead plant tissue. Peat is considered a long-term storage location for nutrients (DeBusk, 1999). One study found that twice as much phosphorus was sequestered in submerged aquatic vegetation as in sediment, but these nutrients had a greater probability to be mobilized as plants decay (Dierberg *et al*., 2002). Dissolved organic phosphorus and insoluble forms of organic and inorganic phosphorus are generally not biologically available until they are transformed into soluble inorganics (Mitsch and Gosselink, 2000). Therefore, both storage of phosphorus within wetlands and the reduction of downstream export of soluble inorganic phosphorus decrease the effective nutrient load of downstream waters and the associated eutrophication.

Nutrient removal in constructed wetlands has been found to follow a seasonal pattern in most temperate conditions. The amount of nitrogen and phosphorus removed depends on the form of the nutrient, type and density of the aquatic plants, nutrient loading rate, and climate. During winter, nutrients sequestered in plants and plankton are released back into the water column upon decomposition (USEPA, 1999). Typically, nutrients taken up by plants and microorganisms in dissolved organic forms are returned later in complex organic forms (Tanner, 2001a). Seasonal temperatures also inf uence transformation of nutrients. For example, nitrification is limited by temperature during all seasons when plant gas exchange and oxygen input into the rhizosphere are limited. Denitrification was almost complete in midsummer and was restricted at seasonal temperatures below 15°C in a study conducted on a constructed subsurface horizontal f ow wetland in Germany (Kuschk *et al.*, 2003). Spring and autumn removal efficiencies responded to the nitrogen load in a linear fashion. Efficiencies in winter and summer differed extremely (mean removal rates of 0.15/0.7 g m−2 d−1 [11 percent/53 percent] in January/August) and appear to be independent of the nitrogen load (0.7–1.7 g m−2 d−1) (Kuschk *et al.*, 2003). Wetland treatment systems in Hungary showed that removal performances varied by 40 percent between summer and winter (Szabó *et al*., 2001). Several studies found that temperate regions show a rapid uptake of nutrients in early spring with rising temperatures, which stimulates mineralization of organic matter accumulated over the previous winter (Tanner, 2001a).

Although several studies demonstrated seasonal inf uences in water quality performance, a study of constructed wetlands in Florida found no seasonal pattern in phosphorus removal despite f uctuations in air temperature and sunshine (Dierberg *et al*., 2002). Sub-tropical wetlands lack the annual cycle of fall-winter senescence and nutrient release that is characteristic of northern climates. However, this lack of seasonality may add to the long-term stability of sediments and detritus-bound nutrients in sub-tropical regions. Another regional characteristic found in Florida, but applicable to other similar areas, is the high level of calcium and high alkalinity in runoff. This regional condition of runoff allowed more phosphorus to be sequestered by co-precipitation with calcium carbonate (Dierberg *et al*., 2002). These examples illustrate the inf uence regional factors have on nutrient removal performance of wetlands and may explain why wetland nutrient removal performance is better in some regions than in others.

Climate inf uences the amount and timing of nutrient input, as well as nutrient concentration and transformation within wetlands (Mitsch and Gosselink, 2000). Temperature affects growth and productivity of wetland biota. Also, oxygen levels in wetlands f uctuate with temperatures; oxygen saturation is greater at cooler temperatures. Oxygen levels, in turn, affect several nutrient transformation processes. For example, Woodwell and Whitney (1977) found a salt marsh uptake of phosphate in cold months and export of phosphate in warm months. Areas with high precipitation have increased hydrologic inputs, which can dilute nutrient concentration or increase nutrient concentrations if the precipitation picks up nitrogen and phosphorus before entering the wetland through overland or groundwater f ows. A study of several streams throughout the United States found that concentrations of nitrogen and phosphorus increased with precipitation in disturbed watersheds because of increased erosion, but decreased with stream f ow in natural watersheds, presumably because of reduced erosion and increased dilution (Omernik, 1977). Arid regions can concentrate nutrients as water evaporates from wetlands, which leaves increased salts, affecting chemical binding rates and biological diversity. Additionally, groundwater may be more inf uential in arid regions as the subsurface water picks up nutrients within the soil prior to outfalling to wetlands (USEPA, 1993a).

Climate also has considerable effect on the plant and microorganisms growing in wetlands. The quantity and variety of these organisms influence the nutrient transformation and removal within wetlands. For example, temperate wetlands retain more nutrients in the growing season primarily because of the higher microbial and macrophyte productivity. Nutrients stored in biomass can be released back into the water column in the autumn following litter fall and subsequent leaching. This seasonality has application to the concept of using wetlands to reduce downstream nutrient loads. Wetlands can function as sinks for nitrogen and phosphorus in summer, when the biotic community is most productive, which corresponds favorably with the need to reduce summer algae blooms in downstream waters as a result of elevated nutrients (Klopatek, 1978; Lee *et al*., 1975).

Nutrient removal has been shown to be higher in wetlands containing plants, mostly through denitrification and secondarily through plant uptake (Stein *et al*., 2003; Lin *et al*., 2002; Jing *et al*., 2002; Tanner, 2001a). Macrophytes have been found to enhance nutrient removal by assisting solid sedimentation, reducing algae production, improving nutrient uptake, and releasing oxygen (Jing *et al*., 2002; Bavor *et al*., 2001). Studies of surface flow horizontal reed beds in Australia found removal efficiency with plants to be greater than 96 percent for both nitrogen (9.7 milligrams per liter [mg/L]) and phosphorus (0.56 mg/L) and without plants to be 16 percent for nitrogen (1.6 mg/L) and 45 percent for phosphorus (0.26 mg/L) (Huett *et al*., 2005). Another study of constructed wetlands in Taiwan found that planted wetlands removed 80 to 100 percent of ammonium (NH_4)-nitrogen (NH_4-N) (Jing *et al*., 2002). High denitrification rates in the presence of plants has been attributed to a high degree of soil oxidation (Matheson *et al.,* 2002). An assessment of subsurface constructed wetlands found that oxygen transport down to the roots by emergent plants was the prime source of oxygen needed for nitrification (USEPA, 1993a).

Submerged aquatic vegetation communities have been found to exhibit phosphorus removal mechanisms not found in wetlands dominated by emergent macrophytes (Dierberg *et al*., 2002). Constructed wetlands using floating aquatic macrophytes have been used to improve drinking water supplies in Brazil (Elias *et al*., 2001). The submerged plants directly assimilated phosphorus from the water column and mediated the pH so phosphorus co-precipitated with calcium carbonate in soil sediment. Leaves and stems can also act as nucleating sites for co-precipitation. Under high iron and oxygen conditions, phosphorus has been found to co-precipitate on iron oxide as evident from purple plaques observed on roots and stems, contributing to a removal efficiency of 83.6 percent (Jardinier *et al*., 2001). Removal efficiencies for organics, NH_4-N, and orthophosphates were influenced by the health and growth rate of macrophytes (Jing *et al*., 2002).

Even wetlands designed to treat wastewater through subsurface flows showed enhanced nitrogen and initial phosphorus removal when planted versus unplanted wetlands with gravel-bed substrates (Tanner, 2001a). Uptake and storage of nitrogen and phosphorus in live plant biomass accounted for a fraction (3 to 19 percent TN; 3 to 60 percent TP) of the improved performance of planted wetlands. The author suggests that plants primarily facilitate improved nutrient removal indirectly through their effects on other removal processes rather than direct nutrient uptake (Tanner, 2001a). A recent study of nitrogen uptake in the rhizosphere concluded that nitrate (NO_3) uptake by wetland plants may be far more important than previously thought. The modeled calculations showed that substantial quantities of NO_{-3} can be produced in the rhizosphere of wetland plants through nitrification and taken up by the roots under field conditions and that rates of NO_{-3} uptake can be comparable to those of NH_{+4}. In addition, the model showed that rates of denitrification and subsequent loss of nitrogen from the soil remain small even where NO_{-3} production and uptake are considerable (Kirk and Kronzucker, 2005).

Many studies have shown that different species of plants perform better than others at nutrient removal from waste water. Cattails were most efficient at nitrogen removal, and aquatic plants increased phosphorus removal in wetlands constructed to treat saline wastewater in Thailand (Klomjek and Nitisoravut, 2005). Careful consideration should be given to the choice of plant species used for nutrient removal systems. While many species can be desirable and effective for nutrient removal in some regions, those same plants can be undesirable in other regions (Mitsch and Gosselink, 2000) and can often be highly invasive, spreading to and causing problems in nearby aquatic systems. Other species that have shown high rates of nitrogen removal from waste water include *Phragmites* (Mayo and Bigambo, 2005), *Typha angustifolia* (Belmont *et al*., 2004), *Scirpus validus* (Fraser *et al*., 2004), and *Schoenoplectus* (Poach *et al*., 2003). However, some studies found that plant species had little impact on nutrient concentration or removal (Jing *et al*., 2002; Huang *et al*., 2000). A study of constructed wetlands in the Florida Everglades found that species differed in their uptake and accumulation in plant tissue, but it was a minor contributing factor in overall nutrient removal (Dierberg *et al*., 2002). In addition to plants affecting transformation processes, plants also take up nutrients into their tissues. Much of the storage of nutrients in plants occurs in below-ground tissues, particularly in emergent species where up to 90 percent of the plant productivity occurs in below-ground tissues (Tanner, 2001a; Wetzel, 2001). This is particularly true when plants enter maturity and senesce as nutrients are translocated to root tissues for storage until the next growing season. Consequently, the removal of above-ground tissue is often not a practical method for removing nutrients from the wetland (Wetzel, 2001; Matheson *et al.,* 2002). Plant tissue analysis has shown that a single annual harvest of plant material accounted for 10 percent or less of the nitrogen removed from constructed subsurface wetlands. Increased harvest frequency may increase this performance, but would increase the operation costs of the constructed wetland (USEPA, 1993a).

Studies of the effect of hydrologic and hydraulic conditions show inconsistent results. Hydrologic and hydraulic conditions in a wetland can inf uence the efficiency of processes that remove nutrients from water (Jing *et al*., 2002; Sakadevan and Bavor, 1999). Hydraulic residence time was negatively correlated with TN and phosphorus removal in constructed subsurface f ow wetlands (Schulz *et al.*, 2003). $NH+_4$ and total Kjehldahl nitrogen (TKN) concentrations within a wetland decreased exponentially with increased residence time (Huang *et al.,* 2000). TKN is the organically bound nitrogen in a water sample that is released from organic matter through a digestive process before analysis. Knight *et al*. (2000) found that removal of nutrients was a function of inlet concentrations and hydraulic loading rates, but in other studies nutrient removal efficiencies were unaffected by variation in hydraulic loading rates (Lin *et al*., 2002). Dierberg *et al*. (2002) found the greater the residence time, the greater reduction in nutrients.

Ideally, the optimal performance of a constructed wetland can be achieved by affecting the inf ow concentration and residence time. Consideration should be given to designs of constructed wetlands with localized inf ows, which generate a nutrient soil gradient. A study of wetlands used for 40 years to treat wastewater in Florida found that TP in wetlands sediments was significantly correlated with depth and distance from the point of surface water inf ow (White and Reddy, 2003). Nutrient retention has been found to be affected by wetland size relative to the watershed (and therefore retention time), land use of the watershed, any intrusion of groundwater, and the nature of the wetland in terms of its shape and vegetation (Raisin and Mitchell, 1995). An assessment of subsurface constructed wetlands found that the media (e.g., gravel, sand) affected the hydraulic conductivity and, subsequently, the nutrient removal performance. Systems with sandy substrate had low conductivity and, therefore, needed to be larger in size to generate a retention capacity effective at removing nutrients, which requires more land surface for construction and operation (USEPA, 1993a).

3.2 Factors that Affect Nutrient Load Reduction Efficiencies

Wetlands that are undersized compared to the amount of water that will f ow through them are more susceptible to frequent f ushing by storms (which can f ush out nutrients and organic matter) and are therefore not as effective as properly sized wetlands. Wetlands need to be large enough to be able to store the total from the “first f ush,” the first 1 inch of precipitation (Hunt and Doll, 2000). Bass (2000) indicated that current recommendations are that a wetland surface area should be at least 1 percent of the contributing watershed area. However, given that the amount of runoff from a drainage area will vary considerably depending of the amount of impervious area within the watershed, Hunt and Doll (2000) calculated surface areas of wetland ranging from 7 percent for a watershed with a low permeability (curve number [CN]=98)[2] to slightly more than 1 percent for residential areas with fairly clayey soils (CN=60).

This illustrates one limitation of constructed stormwater wetlands relative to other stormwater BMPs: they require a large area of land. Wetland designs can improve the overall performance of the wetland and partially address the problem of stormwater f ows f ushing wetlands by including a high f ow bypass (f ow splitter) that allows larger storms to circumvent the wetland (Hunt and Doll, 2000). In North Carolina, constructed stormwater wetlands have been located on watersheds as small as 4 to 5 acres, but they are most commonly used for larger drainage areas and typically serve watersheds ranging from 15 acres to more than 100 acres.

Geographic position and land use affect the nutrients f owing into wetlands (Mitsch and Gosselink, 2000). The size of the watershed, the steepness or slope of the landscape, soil texture, and variety of topography inf uence these nutrient inputs. The position of the wetland within the landscape, in addition to the climatic situation, inf uence the cycling of nutrients within and through wetlands. For example, tidal salt marshes have significant tidal exchange while closed ombrotrophic bogs have little material exchange except for gaseous matter into and out of the wetland. Upstream wetlands have the ability to affect the amount and form of nutrients f owing into wetlands (e.g., a series of wetlands will produce a different outcome compared to a single wetland). Land uses can affect nutrient inputs by affecting erosion rates, applying fertilizers, modifying hydrologic f ows, and altering buffer features of wetlands. Adjacent land use practices also may impact a wetland’s ability to store nutrients, thereby altering the structure and function of the wetland (Gathumbi *et al.,* 2005). Obvious direct input from sewage eff uent, urban runoff, and industry can have dramatic impacts on nutrient loads within wetlands. Studies of a natural wetland in New Zealand that received sewage oxidation pond eff uent for more than 30 years showed elevated nutrient concentrations in ground and surface water, increased weed invasion and plant growth, and high concentrations of certain heavy metals (Chague-Goff *et al*., 1999).

Anthropogenic sources of nitrogen and/or phosphorus include sewage, fertilizers, animal waste, erosion, industrial discharge, mining, drinking water treatment, synthetic materials, and fossil fuel burning. As previously discussed, both phosphorus and nitrogen are present in wetlands in inorganic and organic forms. Both nutrients are used by living organisms for basic life processes, but too much can be harmful to aquatic environments. The potentially harmful effects associated with anthropogenic enrichment of nutrients are most noticeable in environments where these nutrients are normally in limited supply, such as within surface water bodies (e.g., eutrophication). Nitrogen and phosphorus are often found in higher than natural levels in areas of human activity. Consequently, the negative effects of too much nitrogen

2 *CN reflects the ability of a watershed to store water through initial storage and subsequent infiltration. A high CN indicates a watershed with limited storage capacity.*

and phosphorus are concentrated downstream of these areas, leading to the need to reduce nitrogen and phosphorus within water bodies. Removal of nutrients from water before the water is discharged downstream can reduce the potential for eutrophication; however, upgrades to treatment processes cannot eliminate this potential. For example, sewage treatment typically decreases ammonia discharge, which results in increased NO_3 discharge, but does not address TN discharge concentrations (Murphy, 2005).

Additional studies focusing on the design issues of constructed wetlands are necessary. These studies should look at the impacts of scale and edge effects in research wetlands. Also, the delivery of treatment water at a single point or dispersed delivery and in batches versus continuous f ows should be studied further for modeling and application of constructed wetlands and as treatment BMPs. Longer-term studies are also lacking within the literature. Further study is needed on quantifying and comparing the oxygen release characteristics of different emergent species in response to root-zone treatments and the effect of this release on removal efficiencies (Tanner, 2001a).

3.3 Natural versus Constructed Wetlands

Natural wetlands exist where water inundates land, even seasonally, or groundwater is shallow enough to create hydric soils near the surface, which supports hydrophytic plants adapted to living in water or saturated soils. Constructed wetlands developed to improve water quality are defined as engineered or constructed wetlands that use natural processes involving wetland vegetation, soils, and their associated microbial assemblages to assist in the treating of eff uent or other water sources (USEPA, 2000a). Because constructed wetlands are typically designed specifically for water quality improvement functions, many of the wildlife habitat functions provided by natural wetlands are lacking in constructed wetlands (DeLaney, 1995). A third type of wetland, often referred to as a created wetlands, are often designed to provide wildlife habitat functions similar to natural wetlands as mitigation for project impacts (Hammer, 1996). There are generally two types of constructed wetlands: subsurface and free-water-surface systems (USEPA, 1999; USEPA, 1993a; Hammer, 1989).

Restored and enhanced wetlands are historical, naturally occurring wetlands that have been disturbed through filling, dredging, water elevation changes, plant community alterations, and/or modifications to buffers surrounding the wetland that impact the wetland characteristics or functions. Restoration of disturbed wetlands usually involves rehabilitation of hydrologic conditions and reestablishment of vegetation (Mitsch and Gosselink, 2000). Degraded wetlands offer opportunities for restoration and enhancement through the careful application and operation of them for water quality treatment. However, this approach should only be attempted if the water quality of the wetlands would not be degraded, there was a net benefit to the wetland, and it would promote a return of historic or natural conditions to the wetland (USEPA, 2000a). In natural wetlands with low productivity, nitrogen and phosphorus are often limiting factors, and adding nutrient-rich water can increase productivity (Mitsch and Gosselink, 2000; Ewel and Odum, 1984). Restoring wetlands is an effective strategy for reducing agricultural NPS nutrient discharge. These systems can remove 90 percent to 100 percent of suspended solids, 85 percent to 100 percent of TP, and 80 to 90 percent of TN (DeLaney, 1995). A compilation of data from 60 studies of 57 natural wetlands in 16 countries showed that 80 percent of the wetlands reduced nitrogen loading and 84 percent reduced phosphorus loading. The mean percent change in nutrient load between water entering and exiting the wetlands was 67 percent (SD of 27 percent) for nitrogen and 58 percent (SD of 23 percent) for phosphorus (Fisher and Acreman, 2004).

Constructed wetlands designed to retain nutrients from wastewater can function similarly to natural systems. They have similar physical and biological processes and the operation is more passive and requires minimal operator intervention as compared to WWTPs (USEPA, 2000b). Planning and design considerations for building constructed wetlands have been developed by USEPA (1999). Wetzel (2001) provides a summary of the fundamental processes in natural and constructed wetlands. Both natural and constructed wetlands exhibit plant and microbial metabolism involved in nutrient/pollutant uptake, sequestering, and retention that is highly dynamic on daily, seasonal, and long-term annual scales (Wetzel, 2001; Kadlec and Knight, 1996; Ewel and Odum, 1984). Furthermore, the amount and concentration of nutrient loading inf uence these processes at all scales. Nutrient removal rates have also been shown to be very high in some natural and constructed wetlands. A study of 50 years of treating wastewater by f owing it through existing forested wetlands in the Mississippi Delta showed that nitrogen and phosphorus were reduced by more than 90 percent (Day *et al*., 2004). A constructed wetland in France was reported to have removed 54 to 94 percent of TN from coke plant wastewater (Jardinier *et al*., 2001).

Though there are similarities between natural and constructed wetlands, there are also several differences. Constructed wetlands often vary in the shape and structure from natural wetlands. Often, constructed wetlands are shaped to fit into the landscape with other features such as roads, buildings, or mature vegetation. This “fitting in” can limit the ability to create a natural-looking and -functioning wetland. Many of the studies of constructed wetlands use conveniently-sized plots (e.g., mesocosms) that provide straightforward control of soils, plants, and water levels as well as inf ow and outf ow controls, which ease measurement of water quality parameters (Dierberg *et al*., 2002; Jing *et al*., 2002). Additionally, constructed wetlands often have engineered substrates composed of gravels or artificial liners, which affect the subsurface nutrient removal processes.

Natural wetlands are typically higher in biodiversity, while constructed wetlands are typically planted with a few select plants and occasionally are inoculated with microorganisms (Wetzel, 2001). This greater diversity often allows more light to penetrate deeper into the water, increasing the vertical extent of photosynthesis and survival of microorganism assemblages. The increased species diversity and productivity maximizes nutrient retention, recycling, and storage (Wetzel, 2001).

Guidelines for constructing wetlands produced in 2000 identified more than 600 active projects using constructed wetlands to treat municipal and industrial wastewater, as well as agricultural and stormwater sources (USEPA, 2000a). Using these projects and wetland science, USEPA developed “Guiding Principles for Constructed Treatment Wetlands” to develop wetlands that improve water quality as well as provide wildlife habitat (USEPA, 2000a). The document gives guidance on planning, siting, designing, constructing, operating, maintaining, and monitoring of constructed treatment wetlands. Other guidance documents on constructing wetlands have been developed and provide useful information to consider when constructing wetlands (Davis, 2003; Moshiri, 1993; Cooper and Findlater, 1990; Hammer, 1989 and 1996; Kadlec and Knight, 1996). USEPA also developed two technical assessments of different constructed wetlands: *Free Water Surface Wetlands for Wastewater Treatment: A Technology Assessment* (USEPA, 1993a), and *Subsurface Flow Constructed Wetlands for Wastewater Treatment: A Technology Assessment* (USEPA, 1999). These can help determine the selection and design of an appropriate constructed wetland.

Some recent studies provided additional information on design and performance of constructed wetlands. For example, interspersing open water with emergent vegetation appears to maximize NH_4 removal efficiency (Thullen *et al*., 2002). Adding maerl (calcified seaweed) to a laboratory wetland resulted in 98 percent reduction in phosphorus (Gray *et al*., 2000). Wetzel (2001) suggests that all wetland treatment strategies should maximize physical contact and duration of contact between water and microorganisms and periphyton. Periphyton growing on aquatic vegetation have been found to be significant in their assimilation of nutrients (Dierberg *et al*., 2002). The importance of submerged aquatic vegetation and periphyton in improving constructed wetland performance in removing nutrients was demonstrated in studies in the Florida Everglades (Goforth, 2001). Research also indicates that the uptake and return of nutrients are separated in time and occur on different temporal scales, which should be taken into account during the design and operation of constructed wetlands (Tanner, 2001a). A comparison of subsurface systems found that wetlands performed better at removing ammonia when incorporating three design elements: no algae, longer detention times, and deep root penetration of emergent plants, rather than only one or two elements (USEPA, 1993a).

Even though natural and constructed wetlands have been used for water quality treatment for many years, there are still gaps in knowledge on performance and design factors. Studies are still needed to better understand the chemical and physical characteristics of various nutrient fractions in runoff as well as the nature of nutrients that remain after passage through wetlands (Dierberg *et al*., 2002). Other studies have suggested the need for a widespread measurement program to provide a more detailed evaluation of wastewater treatment systems to identify variability and factors contributing to variability (Szabó *et al*., 2001). The nutrient removal rates and capacity in both natural and constructed wetland systems need further investigation to allow identification and comparison of nutrient removal in a wide spectrum of wetland types, scales, landscape positions, regional climates, geology, and nutrient inputs.

3.3.1 Related Outcomes of Constructed Wetlands

Constructed wetlands designed to treat water high in nutrients generate related beneficial and detrimental outcomes. These outcomes provide additional advantages and disadvantages to using constructed wetlands as BMPs in a WQT program that should be considered when selecting this BMP to generate WQT credits. Knight (1992) provides an overview of the ancillary benefits and potential problems with the use of wetlands for NPS nutrient discharge. These related outcomes are discussed brief y and incorporated with other study findings.

Constructed wetlands can provide many benefits in addition to water quality treatment (Kadlec and Knight, 1996). These benefits include: photosynthetic production; secondary production of fauna, food chain, and habitat diversity; export to adjacent systems; and services to human society such as aesthetics, hunting, recreation, and research (Knight, 1992). One of the key biological benefits of constructed wetlands is their ability to provide habitat for plants and animals. Many plants and animals live in wetlands, and many periodically use wetlands as drinking sources, breeding sites, or foraging areas. For example, a series of shallow ponds constructed to maximize NO_3 removal in California had an average avian specie richness ranging between 65 and 76 species per month, including both common and rare species. Wetlands also provide a food source for animals such as nutria and muskrats; however, these species can consume much of the vegetation and reduce the nutrient removal function of constructed wetland (USEPA, 1999).

A summary of 17 case studies located in 10 states found that constructed wetlands can provide valuable wetland habitat for waterfowl and other wildlife (USEPA, 1993b). However, wildlife can sometimes be detrimental to the nutrient removal efficiency of wetlands. For example, in a constructed wetlands near Chicago, a large number of carp were found foraging and resuspending sediment, thus decreasing the performance of the wetland. These fish had arrived as juveniles in the inf ow and grew up in the wetland. In another example, a wetland constructed to remove nitrogen from municipal wastewater included open water habitat to attract waterfowl. Wintering waterfowl and colonial red-winged blackbird (*Agelaius phoeniceus*) used the open water areas, but contributed a small amount (2.6 percent nitrogen and 7.0 percent phosphorus of mean daily loads from WWTP) to nutrient loading during November through March (Andersen *et al.,* 2003).

Using wetlands for nutrient treatment can have demonstrated additional water resources benefits within the wetland and downstream. The use of a natural forested wetland in the Mississippi Delta for wastewater treatment over 50 years has shown significant sedimentation and resulted in increased accretion rates (Day *et al.,* 2004). The results of the study suggest that the application of nutrient-rich wastewater, and the resulting sedimentation, can also gradually increase wetland elevations and counteract some of the negative effects of sea level rise on coastal wetlands.

Adding nutrient-rich water into natural wetlands has been demonstrated to increase productivity of woody vegetation, measured as stem diameter growth, and growth of herbaceous emergent and aquatic vegetation (Day *et al*., 2004). The additional growth of emergent and aquatic vegetation contributes more to sediment accretion. This sedimentation function also improves downstream habitat. Water typically f ows slowly through both natural and constructed wetlands because of their gentle gradient and vegetation. The slow f ow allows fines to settle out or deposit on vegetation. Consequently, fewer fines are transported downstream, benefiting fish. Fines in streams can fill interstitial spaces within gravel substrates, reducing the quality of spawning success in fish.

In addition to improving fish spawning habitat, constructed wetlands can provide additional benefits by ameliorating f ood waters, storing water for multiple uses, and recharging groundwater (Feierabend, 1989; Slather, 1989; Knight, 1992). Watersheds composed of 5 to 10 percent wetlands are capable of providing a 50 percent reduction in peak f ood period compared to those watersheds that have none. Therefore, constructed wetlands can be valuable in watershed management strategies, especially in areas where wetlands have been lost (DeLaney, 1995). The effectiveness of wetlands is determined in part by the location of each wetland in the watershed. In arid regions, the reuse of wastewater through treatment wetlands can be especially helpful in serving to conserve water, provide habitat, recharge groundwater, and maintain longer instream f ows downstream (USEPA, 2000a).

Wetlands built along shorelines of streams, lakes, and marine environments can help control erosion from f ows, wind, and shoreline uses. The erosion is largely controlled by the rooted vegetation established in the wetland, which disrupts the f ow velocities and binds the soil. Constructed wetlands positioned along shorelines need to be carefully designed, constructed, and maintained to ensure inf ow water is treated by the wetland before discharging to adjacent water bodies (Hammer, 1992).

There are several direct human benefits possible from constructed wetlands. The improvement of water quality by wetlands has been found to benefit human health by reducing disease-causing bacteria and viruses (Jing *et al*., 2002). Wetlands remove toxic chemicals found in wastewater in addition to nutrients. Harvesting of wetland vegetation has been used for the production of methanol (USEPA, 1999). Constructed wetlands with public access and public use provide recreation, research, and educational opportunities. Public education has ancillary benefits of generating support for water quality and watershed protection. Constructed wetlands have been used in combination with other treatment mechanisms to provide safe drinking water (Elias *et al*., 2001).

Even though there are many benefits from constructed wetlands designed to treat water quality, these wetlands can also have detrimental outcomes. For example, the use of farmland to construct a wetland results in a loss of that land for farming or another land use. Constructed wetlands located in other water bodies (i.e., wetland, stream, or lake) or immediately adjacent to natural water bodies can negatively affect the natural water quality or quantity of these water bodies (USEPA, 2000a). This effect depends on the quality of the natural water body and the design of the constructed wetland.

Constructed wetlands that attract wildlife may have a negative consequence. For example, siting a constructed wetland near an airport might attract birds, which present a hazard for airplanes and the birds. Constructed wetlands can also be a hazard to wildlife if they provide large amounts of habitat where many birds of various species can interact and spread diseases. Attraction of wildlife could also lead to increased encounters with domestic animals, leading to direct or indirect harm to both animal groups (USEPA, 1999). As mentioned above, wildlife can negatively affect the nutrient performance of a wetland through direct input of nutrients or remobilization of nutrients. If water input is episodic or seasonal, the high f uctuations in water level and potential drought periods could be detrimental for organisms that reside in the wetland. Constructed wetlands can be directly harmful to organisms if the water quality is poor or even toxic. For example, selenium has been found to bioaccumulate in constructed wetlands, leading to reproductive failure in fish and aquatic birds (Nelson *et al.*, 2000; Lemly and Ohlendorf, 2002).

The building of constructed wetlands requires disturbance of soil and vegetation. Disturbed areas are prime locations for colonization by invasive plant species, especially if sources are nearby. Additionally, nutrient loading of wetlands can result in a shift in plant species assemblages, often seen as an increase in weed invasion at the point of eff uent discharge (Chague-Goff *et al.,* 1999). Consequently, constructed wetlands can provide habitat and opportunity for spreading invasive species.

Public health and safety may be compromised by constructed wetlands if they are not designed and maintained carefully. Wetlands can have odors that are unpleasant for neighboring communities. Odors in constructed wetlands are typically associated with high organic loadings, especially near the inlet. Also, without safeguards, wetlands can pose a safety hazard to visitors to the wetland. Constructed wetlands used to treat wastewater need to prevent human contact with the untreated water, which could carry pathogens harmful to human health (USEPA, 1999). In some areas of the country, dangerous reptiles, including poisonous snakes and alligators, could be attracted to constructed wetlands. A USEPA study is examining if treatment wetlands are more or less likely to create risks to wildlife species than adjacent natural wetlands (USEPA, 1999).

Another species attracted to wetlands that can be a nuisance or harmful to humans is mosquitoes. Studies of mosquitoes have concluded that the number of breeding mosquitoes in treatment wetlands is not higher than in adjacent natural wetlands (Crites *et al*., 1995). Controlling vegetation to create dispersed open water patches can result in reduced mosquito populations by limiting mosquito refuge areas and increasing predation areas (Thullen *et al*., 2002). However, another study found that vegetation management within constructed wetlands conducted in autumn to stimulate denitrification correlated with higher mosquito abundance than control wetlands lacking management (Walton and Jiannino, 2005). According to a USEPA fact sheet (2004), as long as wetlands function as healthy ecosystems—i.e., are able to sustain mosquito-eating fish, amphibians, birds, and insects—they are not uncontrolled breeding grounds for mosquitoes. In fact, it was found that mosquito habitat was reduced by almost 100 percent and the *Culex* species of mosquito almost eliminated after a degraded wetland no longer requires mosquito control measures (USEPA, 2004).

There are also potential negative impacts to air from constructed wetlands. Denitrification process within microbes that occur in wetlands converts NO_3 to N_2O, which is released to the atmosphere and has negative effects on local ground-level ozone (DeBusk, 1999). This process occurs in anaerobic conditions, typically below the soil surface. A study of constructed wastewater treatment wetlands in Sweden showed that N_2O emissions varied seasonally during two years of measurements: large spatial and temporal variations were measured in N_2O flux; the largest positive flux of N_2O occurred in October, and the smallest positive flux in July (Johansson *et al*., 2003). The release of CH_4 gas is also a negative outcome of denitrification (Wetzel, 2001). CH_4 gas emissions from wetlands can contribute to local odor issues and add to greenhouse gas levels. Emissions of greenhouse gases (CH_4 and CO_2) were measured throughout an annual cycle and shown to be positively correlated with water temperature in shallow wetland ponds constructed for nitrogen removal (Stadmark and Leonardson, 2005). CH_4 production was most pronounced from May to September when NO_3 concentrations were low. The study concludes that constructed nutrient removal ponds emit greenhouse gases comparable to lakes in the temperate region.

Knight (1992) provides guidance on optimizing the appropriate ancillary benefits and avoiding undesirable side effects while achieving primary nutrient control goals. Many of the benefits and problems with constructed wetlands can be addressed during the planning and designing process. Maintenance following construction of the wetland is also important in prolonging and enhancing the nitrogen and phosphorus removal efficiency and ancillary benefits, while minimizing detrimental outcomes. Thus, the design for constructed wetlands needs to provide access for maintenance.

There are several techniques to improving nutrient removal. For example, partial nitrification of swine waste water prior to discharge to a constructed wetland increased TN removal rates (Poach *et al*., 2003). Another study found that adding iron to the substrate significantly improved phosphorus retention (Cerezo *et al*., 2001). A model showed that increasing nitrification rates in the summer and denitrification rates in the winter would improve nitrogen removal efficiencies. This might be accomplished by increasing carbon supply in winter (Gerke *et al*., 2001).

The selection of the appropriate plants for constructed wetlands affects the performance and maintenance of the wetland. Floating aquatic systems are more affected by pests and cold temperatures and are more expensive to construct and operate than surface-flow systems planted with emergent plants (Payne and Knight, 1997; Hunt and Poach, 2001).

Common plant species used as emergents include bulrushes (*Scirpus sp*.), cattails (*Typha sp*.), and rushes (*Juncus sp*.). These plants are important in transporting oxygen from the leaves and stems to roots, providing an oxidized microenvironment in the typically anaerobic root zone of wetlands (Armstrong, 1964).

The juxtaposition of aerobic and anaerobic zones at the soil-water interface is important for nitrification when ammonia is transformed into NO_3 (Hunt and Poach, 2001). Thus, the amount of oxygen reaching the root zone affects the rate of nitrification. Different plant species transport oxygen at different rates to this zone; therefore, plant selection affects the performance of constructed wetlands at treating nutrients. For example, bulrushes have higher rates of oxygen transport than cattails (Reddy *et al*., 1989; Szögi *et al*., 1994), and the sediment around bulrush roots was aerobic 30 percent of the time versus 0 percent of the time around cattails (Szögi *et al*., 2004). Even so, Wetzel (2001) suggests that rooted emergent plants cannot be expected to aerate saturated sediments because the function of translocating oxygen to the roots is to support the metabolic needs of the root tissues, not to oxidize the sediments.

Although the results of some of the studies cited above suggest that certain plants may transport excess oxygen down to the sediments, if very high levels of nitrogen removal are required from a treatment wetland, procedures that increase oxidation of wastewater prior to entering the wetlands or designs to include open water areas might be needed to increase nutrient removal efficiency (Hunt and Poach, 2001).

Removing accumulated emergent biomass and physically limiting the area available for vegetation reestablishment significantly improved the ammonia removal efficiency. Limiting emergent plants mimics early successional patterns with actively growing plants and results in interspersed open water, which also reduces mosquito populations by increasing predation areas (Thullen *et al*., 2002). Harvesting shoots may not be important for long-term nitrogen removal because most of the nitrogen is removed through denitrification (Wetzel, 2001; Matheson *et al*., 2002). Tanner (2001b) found that sediment containing high organic matter accumulated twice the nitrogen and six times the phosphorus than live and dead plant tissue (Tanner *et al*, 1998). Therefore, harvesting the above-ground portions of emergent vegetation might provide only a small contribution to long-term removal of nitrogen and phosphorus from the system.

Because constructed wetlands mimic natural systems, they are, by design, naturally functioning, passive, and require limited operational maintenance. However, the imitation of natural systems does not eliminate the need for maintenance of constructed wetlands. The most critical element of maintenance is the quick identification and action when water level adjustments are needed (USEPA, 2000b). Water level affects many of the processes occurring within the wetland and the survival of aquatic organisms. Regular inspections are fundamental to identifying problems and taking corrective actions, such as adjusting weirs or other water level control features (Kadlec and Knight, 1996).

Constructed wetlands have maintenance requirements similar to stormwater ponds, including hydraulic water and depth control, inlet/outlet structure cleaning, grass mowing of berms, inspection of berm integrity, wetland vegetation management, disease vector (e.g., mosquito) control, and accumulated sediment/organic matter management. Subsurface systems are prone to clogging and are limited in function by oxygen diffusion (USEPA, 1993a). Surface systems may need extraction of built up sediments or vegetation that block f ows (USEPA, 1999). Inspections may identify the need to eliminate or control invasive or nuisance species (USEPA, 2000a). Sprinklers have been used successfully to control adult mosquito populations in constructed wetlands because the sprinklers disrupt the water surface, affecting ovipositioning (Epibare *et al*., 1993).

Review of the related outcomes of constructed wetlands identified several research needs. The quantitative magnitude of related benefits and detriments may vary greatly from one system to another (Knight, 1992). Therefore, related outcomes need to be quantified and compared to different designs, regional variation, human values, etc. For example, studies are lacking on odor associated with constructed wetlands used for water quality treatment, especially in comparison with natural wetlands (USEPA, 1999). The causes, controls, and magnitude of odors as well as their community acceptance would benefit from research.

There is additional need to monitor reference wetlands to compare performance of constructed wetlands and impacts of external factors on wetlands. Monitoring should also include surrounding area as well as the constructed wetland. The design and management of constructed wetlands lack complete understanding and incorporation of problems of channelization, altered microhydrology at the spatial scale of microbes, and assimilation versus physical absorptive retention (Wetzel, 2001). More research is needed on the temporal nature of nutrient removal by constructed wetlands. For example, one study found nitrogen removal efficiency dropped from 79 to 21 percent in one year (Tanner *et al*., 2005). Removal efficiencies also dropped between the first and second year in experimental mesocosms (Hench *et al*., 2003). These changes in removal efficiency could be attributed to seasonality, wetlands maturity rates, or regional factors. The use of constructed wetlands for trading programs could benefit from additional planning and understanding about the long-term performance and fate of constructed wetlands.

3.4 Modeling Nitrogen and Phosphorus Removal by Wetlands

Modeling is used to quantify the performance of processes and to attempt to optimize this performance. Models are useful for acquiring information about performance when actual measurement is prohibitively expensive (Johansson *et al*., 2004). The benefits of accurate models include improved designs, reduced monitoring, and predictability of performance. This predictability could be used to define credits in a market-based WQT program. A predictive model for constructed wetlands should be able to describe and predict wetland hydraulics, because this directly affects the treatment performance of a wetland according to basic water quality modeling such as the k-C* model (Bojcevska, 2005; Persson, 2005; Kadlec, 2000; Persson *et al*., 1999; Wong and Geiger, 1997; Kadlec and Knight, 1996).

Although the physical and biological processes that drive wetland systems are complex, many mathematical models have been developed to simulate nutrient removal in wetlands. Many of these models were developed by accounting for hydrologic conditions and nutrient dynamics. A mathematical model was developed from studies of lowland rice fields and can be used to assess the extent of absorption from the rhizosphere by wetland plants growing in f ooded soil, incorporating important plant and soil processes (Kirk and Kronzucker, 2005). McBride and Tanner (1999) developed a mathematical model to simulate patterns of nitrogen removal that were observed in experimental studies of constructed wetlands treating NH_4-rich water. Brown (1988) developed a simulation model to predict water quality of outf ow water from natural and constructed wetlands. The model requires data input for wetland type, discharge rate, and concentration of nutrients in surface water inf ow (Brown, 1988). Another mathematical model that simulates wetland hydrology and nutrient-driven interactions between wastewater and wetlands was tested by comparing simulations with data from a wastewater treatment facility (WTF) (Kadlec and Hammer, 1988). The simulation accurately predicted solute concentrations, biomass growth patterns, changes in the litter pool, and soil accretion rates. Another two-part model was developed by Dorge (1994) that contains a hydrological submodel and a more complex biological submodel. The model was developed to determine the retention and removal of nitrogen in wetlands as water f ows from cultivated agricultural land through wetlands to aquatic systems. The model can be used to describe the transport and turnover of nitrogen from fertilization through soil and groundwater to aquatic systems (Dorge, 1994).

Some models have focused specifically on plant uptake of nutrients (Langergraber, 2001; Mankin and Fynn 1996; Romero *et al*., 1999; Wegehenkel, 2000). Langergraber (2001) developed a model (CW2D) to simulate plant uptake of nutrients in constructed subsurface f ow wetlands relative to water uptake. The model was tested with indoor pilot-scale constructed wetlands. Langergraber (2005) tested the CW2D model for the portion of nutrient removal attributable to plant uptake and concluded that it is possible to simulate plant uptake of nutrients in constructed wetlands with a model that links nutrient uptake with water uptake. Another model, HYDRUS-2D, also models nutrient uptake by plants coupled with water uptake (Simunek *et al*., 1999). A mass balance method was used to quantify the performance of nutrient storage systems in an experimental artificial wetland (Breen, 1990). In this simulation, hydrologic design to maximize wastewater-root zone contact was determined to be important for treatment performance. Furthermore, uptake by plants was found to be responsible for most of the nutrient removal, and plant biomass was determined to be the primary nutrient storage mechanism. Other studies that included field measurements of nutrient uptake in constructed wetlands often come up with the opposite result; plant uptake is a relatively small component of total nutrient uptake compared to microbial processing (Hamersley *et al.,* 2001; Lin *et al.,* 2002; Stein *et al.,* 2003).

Simulations of natural wetlands have also been modeled. A model was developed specifically for riverine wetlands to describe the interaction and processing of carbon, nitrogen, and phosphorus (van der Peijl and Verhoeven, 1999). The simulation results showed a good fit to data collected on riverine wetlands in southwestern England. In a later test of the model to study nutrient enrichment of a riverine wetland, results diverged from the field studies when the simulations predicted a far greater role for nitrogen as limiting factor than the field experiments (van der Peijl *et al*., 2000). The lack of agreement between the simulation and the field experiments was attributed to differences in the environmental conditions (e.g., weather and area measurements) between the field experiment and the computer simulation.

Field-scale simulation models have recently been practiced instead of intensely and expensively surveying farms or conducting field trials for the myriad of conditions in a watershed (Johansson *et al*., 2004). The advantage of field-scale models is that they account for variability in land cover, soil, tillage, and drainage practices. An example of this type of model is the Agricultural Drainage and Pesticide Transport (ADAPT) model. This model simulates the nutrient loads and crop yields resulting from alternative phosphorus BMPs using variable management practices (e.g., crop choice, fertilizer use) and climatological data (Johansson *et al*., 2004).

Watershed modeling has been used to predict nutrient loadings (Arheimer and Wittgren, 2002; Gowda *et al*., 1998). For example, a study in Eastern Europe between Estonia and Russia used a large-scale geographic information system (GIS)-based nutrient transport model over a 15-year period to model the change in nutrient levels caused by reduced agriculture experienced by the region since the restructuring of the former USSR (Mourad and van der Perk, 2004). The study applied the modeling approach developed by De Wit (1999, 2001), the PolFlow model, which used large-scale, spatially variable estimates of sources, transport, and decay of TN and TP over five-year periods. The model consists of three steps: estimating both diffuse (i.e., nonpoint) and PS emissions; calculating long-term hydrological f uxes; and modeling the transport of emitted nutrients through the soil, groundwater, and surface network.

Results from applying the PolFlow model were compared to measured loads and were found to coincide reasonably well with one river and overestimate loadings for another with a smaller drainage basin. In the model, nutrient retention within a drainage basin is simply modeled using a transport fraction factor that is determined by slope and discharge. The study found that modeling was complicated by the transfer of nutrients from nonpoint emissions, which is strongly governed by the retention in and periodic release from storages such as root zone, tile drains, ditches, channels, substrates, f oodplains, etc. Future research is needed to refine the quantification of this nutrient transport fraction. Improvement to modeling nonpoint emissions was suggested by increasing knowledge about the spatial and temporal distribution of various nutrient storage and f uxes along pathways between the soil surface and water bodies (Mourad and van der Perk, 2004).

In north Georgia, watershed-scale modeling is being used to estimate phosphorus loads for different NPS agricultural practices. The Soil Water Assessment Tool (SWAT), based on the USEPA Better Assessment Science Integrating Point and Nonpoint Sources software, is used for rural watersheds and can estimate phosphorus loads by calculating soil loss. The model is calibrated using field samples and local watershed data. Calibration is conducted for two reasons: to determine the parameter values that characterize the general hydrology of the watershed, and to find the parameter values that describe phosphorus and sediment losses from agricultural sources and the effect of BMPs (River Basin Center [RBC], 2003).

The DUFLOW model was developed in The Netherlands for simulating one-dimensional unsteady flow and water quality in open channel systems (EDS, 1998). This model allows for the modeling of pollutant transport and defines processes and pollutant interactions. A similar model was developed and applied to wetlands surrounding Lake Victoria, Tanzania, to simulate the buffering process of wetlands and the capacity of individual natural wetlands to absorb sediments, nutrients, and pollutants. This model estimated the impacts of inputs on water quality, quantity, and accumulation rates in permanent fringe wetland and seasonal floodplain wetlands. This model included both nitrogen and phosphorus compounds and 28 different parameters. The application of the model showed that there was seasonal flow from the lake to the wetlands (Mwanuzi *et al*., 2003).

A study in southwest Sweden was conducted to examine the applicability of the GLEAMS model to simulate the drainage discharge and nitrogen and phosphorus concentrations in the discharge water from a clay field with drain tiles (Shirmohammadi *et al*., 1998). The results indicated that GLEAMS was capable of simulating reduction of NO_3 and dissolved phosphorus losses reasonably well, but there were no algorithms to simulate the particulate phosphorus losses via drain tiles. Therefore, a submodel, "PARTLE," was developed and tested. These two models, combined, provided reasonable estimates of particulate phosphorus loss via drainage through soil. The study concluded that considering the impact of preferential flow and the ratio of annual drainage discharge to annual precipitation is necessary for proper predictions of particulate phosphorus in structured soils.

Modeling fate and behavior of pollutants requires simulation of both transport and controlling processes such as sedimentation, biomass uptake, sorption, etc. (Mwanuzi *et al*., 2003). Modeling nitrogen flux in the lower Mississippi River has been investigated by McIsaac *et al*. (2002). One model they examined accounted for 85 percent of the variation in observed annual NO_3 flux, but tended to underestimate high NO_3 flux and overestimate low NO_3 flux. Another model that used water yield and net anthropogenic nitrogen inputs (NANI) accounted for 95 percent of the variation in riverine nitrogen flux. The NANI approach accounted for nitrogen harvested in crops and assumed that crop harvest in excess of the nutritional needs of the humans and livestock in the basin would be exported from the basin. The U.S. White House Committee on Natural Resources and Environment (CENR) developed a more comprehensive nitrogen budget that included estimates of ammonia volatilization, denitrification, and exchanges with soil organic matter. The residual nitrogen in the CENR budget was weakly and negatively correlated with observed riverine NO_3 flux. When the CENR nitrogen budget was modified by assuming that soil organic nitrogen levels had been relatively constant, and ammonia volatilization losses were redeposited within the basin, the trend of residual nitrogen closely matched temporal variation in NANI and was positively correlated with riverine NO_3 flux in the lower Mississippi River (McIsaac *et al*., 2002).

Crop yield simulation models that incorporate spatial information may apply to modeling nutrient removal in constructed wetlands. Many of these models predict nutrient cycling such as nitrogen and phosphorus fertilization, nutrient transformations, crop uptake, and nutrient movement (Priya and Shibasaki, 2001).

Typically, robust and general models combine both empirical and mechanistic modeling. To gather large amounts of data for empirical modeling, large databases have been developed. One of the most comprehensive summarization efforts to date was the development of the NADB, funded by USEPA (USEPA, 1994). Two versions of the database were ultimately distributed. Version 1, completed in 1994, used an MS®-DOS database system known as Dbase III and was the most widely distributed version. Version 2 of the NADB was built upon an MS® Windows Access database engine. Collected data is analyzed using regression to determine relationships between variables. However, regression does not necessarily indicate causality; thus, spurious relationships can be modeled. Research databases have been used to validate and modify computer models (Humboldt University, 2000).

The first NADB database fell short of meeting its goal of providing sufficient information to optimize the design of treatment wetlands (USEPA, 1999). The bulk of the entries in the revised USEPA-sponsored database (NADB Version II) have been placed into a new database called the Treatment Wetland Database (TWDB). This web-based database adds many additional treatment wetlands to the USEPA-revised database. While the emphasis is on constructed wetlands, natural wetlands are also included in the TWDB database (Humboldt University, 2000).

Rigorous models for constructed wetland systems need to be developed by designing a comprehensive series of iterative studies, collecting data based on quality-controlled specifications, and analyzing the relationships between design features, environmental parameters, and performance. An assessment of current modeling efforts suggests that an effective plan is needed for the design of studies that will provide a comprehensive understanding of the processes that occur within constructed wetlands. The study design should include extensive, quality-assured, transect data at numerous selected sites to capture spatial variation over an extended period of time to identify temporal variation. Using existing mathematical models of wetlands processes combined with the study data, an iterative model of complex systems can be developed and used (USEPA, 2000b).

Modeling constructed wetlands is complicated by the complexity of the reaction mechanisms within these systems, the difficulty in charactering the constituents within the inflow water, and the accountability of influential physical and external factors. Additional challenges include the ability to scale up, shortcomings in analytical and sampling methods, and the capacity to verify models with long-term monitoring (USEPA, 2000b). Modeling is also problematic because wetlands are highly ephemeral in capabilities and efficiencies for uptake and especially biologically-mediated retention of nutrients and pollutants (Wetzel, 2001). Proper model selection is one of the most important steps in any modeling exercise (Priya and Shibasaki, 2001). Many of the current design models for constructed wetlands rely on the assumptions of steady-state water flow conditions and first-order decay of pollutants. Studies have suggested that this is not representative of field conditions (Kadlec and Knight, 1996; Persson *et al*., 1999; Persson and Wittgren, 2004). Thus, there is a need for more experimental data to further define how hydraulic patterns are affected under different environmental conditions, both spatial and temporal.

Further research is needed to improve nutrient models, including detailed hydraulic investigations of full-scale wetlands, simulations of outdoor constructed wetland systems, investigation of plant uptake models, improving the simulation tool by accounting for substrate clogging processes, and developing experimental techniques to measure model parameters (Langergraber, 2003). More work is needed to adequately account for field environmental conditions in computer simulations (van der Peijl *et al*., 2000). Modeling nutrient removal by wetlands should account for delays in nutrient flow pathways through groundwater. There are temporal lags in groundwater flow depending on the size of the aquifer extent and recharge zone, as well as soil type and geology. Consequently, land-use management practices to reduce nutrient loading to a watershed might not result in water-quality improvements for many years, especially if implemented on land far from streams (Wayland *et al*., 2002).

Additional incorporation into models of microbial and hydrological influences on nutrient uptake could improve the predictability of nutrient reductions. Models tend to underestimate that most nutrients from influent sources are assimilated directly by microbiota (i.e., bacteria, algae, fungi) rather than plants and are intensively recycled amongst these microbial communities, which cover all wetted surface in aquatic ecosystems (Wetzel, 2001). Channelization and variability in flow velocity are among the greatest limitations to maximizing retention capacities of nutrients in wetlands (Wetzel, 2001). If these channels and flow patterns are not included in models, then the predictability of the models is hindered by the inadequate consideration of these patterns and their effect on absorption/adsorption rates. Advances in understanding the hydraulic performance in wetlands can be gained by studying water flow patterns or hydraulic residence time distributions obtained from tracer experiments (Persson, 2005).

3.5 Defining Nutrient Load Reduction Credits

A comprehensive review of WQT in the United States identified 40 trading initiatives in 17 states, 29 of which specifically cover nitrogen or phosphorus (Breetz *et al*., 2004). According to the information on WQT programs compiled by Breetz *et al*. (2004), potential NPS WQT partners include: new or expanding WWTPs trading with stormwater BMP retrofits, street sweeping, land reclamation, surplus reductions from existing WWTPs, diverted flow from existing WWTPs, conversion from surface to subsurface discharges, removal of poorly functioning septic systems, or wetland restoration.

The service area for WQT programs (i.e., the area in which trades are allowed) is most often defined by a watershed or sub-basin boundary. A trading program in New York allowed trades only within the same basin, with the exception of one WWTP that received credit for reduction in upstream phosphorus in a basin hydrologically connected to the basin of discharge (Breetz *et al*., 2004). Establishing a trading service area can be further complicated by political boundaries, particularly in watersheds that cross state boundaries. Further division of hydrologically-related boundaries into trading zones may be necessary in some area because of non-uniform mixing of nutrients in water bodies (Kramer, 2000). Credits are often restricted to sources upstream from the point of discharge (Breetz *et al*., 2004).

Building sufficient credit inventory to make a trading program cost-effective can be accomplished in areas that have certain conditions favorable for the establishment of WQT programs. Favorable conditions usually include a wide variation in PS control costs, a large number of PSs, and the availability of low-cost NPS reductions (Kramer, 2000). The seasonality of NPS reductions through implementation of BMPs is also an important factor to consider. The extent to which the spatial and temporal patterns in wetland (or other BMP) nutrient removal performance match the spatial and temporal patterns in load reductions needed by the PSs can determine whether NPS reductions would be appropriate to offset PS discharges (Crumpton, 2006). Further organizational details that are required for a successful trading program are outlined by Stavins and Whitehead (1996). These details include clearly defining responsibility for total discharge; defining trading area; establishing legal authority for trades through rulemaking, legislation, and NPDES permits; monitoring or statistical models to verify compliance; establishing procedures to reduce the costs of identifying potential trading partners, negotiating trades, and program administration; encouraging public involvement to help speed the regulatory process; and regular evaluation of the program for overall efficiency.

Most BMPs used in WQT programs are general and are applicable to many agricultural operations; a few are specific to certain farming activities. Example BMPs used in WQT programs include: livestock exclusions, buffer strips, constructed wetlands, wet ponds, alternative surface tile inlets, cover cropping, roof gutters, filter walls and filter strips, manure storage pits, conservation tillage, runoff control systems, settling basins, concrete barnyards, diversions, underground outlets, livestock exclusion rotational grazing, wetland restoration, land set-asides, nitrogen application restriction, manure incorporation, sediment reductions through land acquisition, conservation easements, streambank stabilization, development of silt basins, dry dams, terraces, grassed waterways, filter strips, and grade control structures (Breetz *et al*., 2004; Kramer, 2000).

Determining credit value for NPS operations is primarily based on getting agency concurrence of acceptable BMPs that reduce nitrogen and phosphorus loading. Some agencies have developed a list of BMPs that are eligible to be used in WQT programs (Idaho Department of Environmental Quality [IDEQ], 2003). The nutrient reductions from these BMPs are usually required to be surplus, quantifiable, permanent, and enforceable.

Creating credits can be difficult in watersheds where agricultural sources are significant contributors to nutrient loads. A common assumption is that agriculture can be a primary supplier of these credits; however, the willingness of farmers to participate in such programs can be problematic for several reasons. Often, trading guidelines prohibit farmers from selling credits when making legally required (e.g., by state regulation) land management changes[3] or for which the farmer has already been paid (e.g., green payments). These prohibitions reduce the ability of farmers to supply low-cost credits. Because they require farmers create credits by implementing BMPs in addition to current practices and then demonstrate that the BMPs do indeed reduce discharge levels (King, 2005). Many BMPs do not show direct improvements and are not easily validated. Rahr, LBR and North Carolina have skirted this issue by assigning typical performance values to specific BMPs. Applying additional BMPs and validating their effectiveness can be a risky endeavor for credit producers because there is no guarantee that the time and money spent will generate more credits.

The need to establish a baseline nutrient load and show reduced discharge levels after BMP implementation creates two additional obstacles for farmers considering supplying credits. First, in order to establish the baseline to quantify marketable credits, an outside party must determine what nutrient-reducing land management practices and/or BMPs farmers have already implemented.) This evaluation is something most farmers are leery about because it could generate questions regarding their justification for green payments or repercussions related to the legality of their land use practices with respect to state requirements. Second, farmers know that their NPS nutrient discharge is currently not regulated as much as PS discharge because NPSs can be difficult to measure, are weather-dependent, and can be costly to control. By showing that they can create baseline information and then reduce their discharges below baseline, they are actually demonstrating that NPS discharge is measurable and that perhaps it should be regulated the same as PS discharge (King, 2005). Farmers are reluctant to participate in a program that could lead to additional regulatory controls over their activities. The LBR program attempts to sidestep this issue through the approach for calculating nutrient credits. The baseline load of a NPS is first determined using the USDA-NRCS Surface Irrigation Soil Loss (SISL) model. Credits generated by a BMP are calculated by subtracting the individual NPS share of nutrient reduction required in the TMDL from the total nutrient reduction created by a BMP (baseline load multiplied by the BMP effectiveness ratio [Breetz *et al*., 2004]).

3.5.1 Measuring Nutrient Removal Performance

Estimating or quantifying existing NPS nutrient loads is necessary for calculating credits and for providing a baseline to measure performance. Methods for measuring baseline conditions and performance of NPS nutrient reduction efforts are highly dependent on the type of activity being conducted and the associated land use practices. Credits have been granted for reductions in nutrient loads achieved through livestock exclusion, stabilization of eroding stream banks, conversion of farmland back to f oodplain, and vegetation restoration. These activities result in reductions in sediment and soil loss as well as the associated nutrient reductions (Fang and Easter, 2003). Other programs have granted credits for voluntary reductions as quantified by a "qualified soil and water conservation professional" according to standardized procedures (Breetz *et al*., 2004).

Where nutrient reduction data are limited and models contain uncertainties, as is currently the case of constructed wetlands on a watershed scale, measurements of nutrient reductions can be taken to determine credits. Performance can be measured as power (nutrient mass removed over time) or efficiency (nutrient fraction removed over time). Direct measurement of nutrient reduction performance of a constructed wetland requires measuring the difference in nutrient concentration between water inf ows to and outf ows from the wetland. The amount of actual nutrient reduction can be measured using grab samples taken during the BMP operation. In the LBR WQT program, the measurement schedule is determined in the trading contract for specific watershed-scale BMPs and regulatory guidance (ISCC, 2002).

3 State land management requirements are relatively rare. North Carolina is an example of a state with land management requirements in some watersheds.

Measuring the nutrient removal performance of a BMP has advantages and disadvantages. An advantage of measuring over calculating nutrient reduction is that it diminishes uncertainties, especially in terms of modeling nutrient loss, nutrient removal by the BMP, and final nutrient loading in downstream water bodies. A disadvantage of measuring the effectiveness of nutrient reduction is that it is very difficult and time-consuming in natural and restored wetlands because the inlets and outlets often extend over relatively broad areas. It is much easier to measure the effectiveness of constructed wetlands than natural wetlands because they can be designed with limited inlets, and outlets are often confined in order to control water levels. The difference in concentrations of phosphorus, nitrogen, and other water quality parameters of interest can be measured at the inlet and outlet, and can be taken as a direct measure of nutrient removal efficiency of the wetland. However, measurement approaches need to account for diurnal, seasonal, and spatial variability in nutrient retention efficiency (Wetzel, 2001). A review of 60 wetland studies showed that the duration and frequency of sampling, as well as which nutrient forms were analyzed, inf uenced in part whether the wetland appeared to reduce or increase nutrient loading (Fisher and Acreman, 2004). Studies that included frequent sampling during high-f ow events, or that were conducted for more than one year, were more likely to indicate that the wetland increased nutrient loading, which is the opposite of the expected result. Nutrients can be f ushed out of wetlands during high-f ow events, which results in an increase of nutrients contained in water exiting a wetland. Wetland design can be used to mitigate or prevent this from happening. Measurements need to be taken throughout the year in order to capture the variations in removal efficiency that wetlands experience over time and seasons (Fisher and Acreman, 2004).

In addition to temporal factors, removal efficiency can vary depending on the position the wetland has in the landscape and in the watershed. For example, wetlands high in the watershed may have limited opportunity to intercept nutrients, and wetlands low in the watershed may have a f ow-through rate that limits efficiency. Efficiency is also affected by the geologic and ecologic conditions in the wetland, where different plant species or vegetation structure vary in their ability to inf uence nutrient removal (Mitsch and Gosselink, 2000). As described in the following section, WQT ratios can be designed to account for the location of a BMP within a watershed.

3.5.2 Modeling and Calculating Nutrient Removal

Credits generated by implementation of BMPs can be modeled or calculated if it is too costly or infeasible to measure the actual performance of the BMP. The first step in calculating credits is to determine the amount of nutrients produced at a location. For example, to estimate the current phosphorus loads from cropland, formulas, such as the Revised Universal Soil Loss Equation and SISL Equation, are used as the most accurate and simple method to estimate soil loss from surface-irrigated cropland (ISCC, 2002; ETN, 2003). These tools can be used to calculate the tons of soil loss per acre per irrigation season. Phosphorus reduction is compared against the phosphorus loads in baseline years used for the TMDL (ISCC, 2002). As another example, reductions in phosphorus loads from cattle exclusion and rotational grazing can be derived by calculating the volume of manure deposited and the associated phosphorus content and delivery ratio (Breetz *et al*., 2004).

Once the nutrient load has been calculated, the nutrient reduction from BMPs is needed to generate credits. One method of calculating potential nutrient reduction is by estimating the average nutrient load reduction associated with a BMP. Nutrient load reductions achieved through agricultural BMPs can also be estimated using field-scale water management simulation models such as the ADAPT model. The ADAPT model can be used to model erosion and sediment transport, which allows for an estimate of phosphorus load reductions from cover cropping, tillage practices, fertilizer applications, crop rotation systems, and planting/harvest dates (Fang and Easter, 2003).

When modeling or calculations are used to estimate nutrient reductions, WQT programs tend to apply a discount to compensate for the uncertainty associated with the effectiveness of the BMP, the accuracy of the modeling results, and geographic variations in nutrient loads and environmental benefits. The multiplier, which is often expressed as a ratio (e.g., 2.1:1 is the trading ratio used by the Neuse River Basin WQT program), is used by WQT programs to reduce the number of transferable credits generated by a BMP. The trading ratio is designed to account for the level of uncertainty associated with the methodology selected to calculate credits, and it is also often established for WQT between NPSs and PSs to include a margin of safety to account for uncertainty in the determination of load reduction (Kramer, 2000).

Credits are also sometimes discounted using delivery ratios to account for location of the BMP project versus the location of the nutrient source that is purchasing the credit. Location within the trading service area can affect credit value. Delivery ratios were developed for the LBR program, which vary from 100 percent in riparian areas, to 20 percent within ¼ mile of the receiving water body, to 10 percent at distances greater than ¼ mile from the receiving water body (Breetz *et al*., 2004). Ratio discounts range from 1.1:1 to 3:1. Overall, trading ratios are applied in WQT programs to ensure that water quality in a watershed is protected and trades between sources distributed throughout a watershed result in environmentally equivalent or better outcomes at the point of environmental concern (IDEQ, 2003). To minimize local impacts or hot spots from PSs off-setting some of their nutrient discharges through trades, NPDES permits may place a limit on the total amount of the nutrient discharge the PS may be off set through Another common approach to minimizing the creation of hot spots, requiring prior approval from the organization that administers the trading program or the state WQ regulator ensures the trade does not result in localized impacts to water quality.

Other WQT programs have developed several ratios used in combination to address uncertainties. In Idaho, a River Location Ratio accounts for the transmission loss of phosphorus occurring within the river system. Site Location Factors account for transmission loss due to phosphorus uptake by plants, water reuse, and the portion of phosphorus that will bind with river sediments and settle out. Drainage Delivery Ratios are determined using a linear calculation of phosphorus transmission loss in the subwatershed's main channels (IDEQ, 2003). Additional information on trading ratios is also included in Section 4.3.2.5.

3.5.3 Assessing and Verifying Performance

The performance of BMPs needs to be assessed and verified to ensure a WQT program is successful. In the Idaho WQT program, BMPs are certified as installed according to NRCS and meeting applicable laws and regulations. Once the BMP is certified and operational, phosphorus reduction credits can be generated and traded (IDEQ, 2003). Monitoring is another way to evaluate performance of BMPs. In Idaho they are used to demonstrate that the BMP is designed and maintained properly, and the program guidance requires at least one annual field inspection to evaluate BMP performance. Constructed wetlands are to be evaluated before and during the middle of the season of use (ISCC, 2002). Another program suggests field spot checks should be performed for BMPs with a maintenance life of over one year. The number of checks is determined based on an annual percentage of those BMPs (ETN, 2003).

Although protocols that produce reliable, quantifiable results have been established to monitor discharges from PSs for most industries, similar protocols are not available to measure discharges from NPSs. Generating reliable, long-term monitoring data of NPS discharges is one of the major challenges faced by WQT programs (Breetz *et al*., 2004). Many trading programs do not have systems for monitoring discharges from NPSs because it would be prohibitively expensive and a long monitoring period is required to provide conclusive results (Breetz *et al*., 2004; Jaksch 2000, Fang and Easter 2003). Periodic reviews of BMPs are often used in lieu of quantifiable monitoring. Some programs use a combination of site-specific inspection at 5 to 10 percent of BMPs and continuous water sampling every eight hours at four locations on a sub-watershed scale (Breetz *et al*., 2004).

Models used to determine nutrient loads and nutrient reductions also need to be verified. A common method to verify models is to calibrate them using local data. For example, stream flow conditions are monitored and grab samples are collected to calibrate SWAT for flow and phosphorus removal rates. In addition, background levels of soil phosphorus are determined by soil samples and used to calculate a soil phosphorus extraction coefficient, which is used to calibrate SWAT. Other models can also be calibrated using daily data of groundwater, interflow, and overland flow from different land use and soil combinations. Several years of data are required for accurate calibration (RBC, 2003). Validating models must consider spatial and temporal scales as well as data sources and manipulation (Priya and Shibasaki, 2001). Modeling nutrient fate and transport within a watershed is an extremely complex technical field, and a large volume of information is available on various modeling techniques used in watersheds across the United States. Assessing the various methods being used to model nutrients within a watershed is beyond the scope of this paper, but is an important research need.

3.5.4 Determining the Useful Life of Credits

Many programs establish time limits on the useful life of BMPs, after which it may no longer be effective. The length of time a BMP can be used to generate credits, tends to be a function of how long it tends to be effective at removing nutrients, with a margin of safety added (ETN, 2003). A comprehensive survey of trading initiatives found that structural BMP credits were assigned a 10-year useful life, and non-structural BMP credits were typically good for 3 years (Breetz *et al*., 2004). A BMP's maintenance life and a margin of safety for uncertainties are used to determine the duration of credits (ETN, 2003). Credited reductions are also sometimes limited in time to be contemporaneous with credit use (e.g., the term of a NPDES permit) (Kramer, 2000).

BMPs have been given individual life spans to assure credit buyers that credits would be available and to assure credit sellers that opportunities to market their credits persist for at least the designated life span of the BMP they choose to implement. In some WQT programs, the life span assigned to BMPs reflected the professional judgments of scientists, regulators, and field practitioners. In the LBR case study, constructed wetlands were originally assigned a 5-year life span, but this was increased to 15 years based on discussion within a technical focus group (Koberg, 2006). In the Tar-Pamlico case study, the credit life span for constructed wetlands is currently 10 years. The handling of credits that have been banked, but not used within 10 years, is one of the issues participants in this WQT program are currently working to resolve (Huisman, 2006). More research and discussion are needed to evaluate and determine the ecologically and programmatically functional life spans for constructed wetland BMPs used in WQT programs throughout the United States, the change in BMP performance over this life span, and the relationship of this life span and performance to water quality credit value.